Control of WATER POLLUTION from cropland

Volume II— An overview

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Office of Research and Development
Environmental Protection Agency

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Control of

WATER POLLUTION

from cropland

Volume II — An overview

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CHAPTER 1

INTRODUCTION

B. A. Stewart and D. A. Woolhiser

Agricultural technology is one of the real strengths of the United States. Although the population has increased steadily, food and fiber production has met the domestic needs and has also provided substantial amounts for export, which is so important to the U.S. trade balance. Fertilizers and pesticides have played a major role in this accomplishment because the acreage of cropland has changed little in the last 45 years–agricultural chemicals and other technological inputs have been substituted for land.

The marvels of agricultural technology have not gone unchallenged. Much of the blame for polluted streams and lakes is often placed on agricultural activities. Some groups and individuals have even called for a total ban on the use of agricultural chemicals. At the other extreme, there are those who claim that the use of chemicals has not had any adverse effect on the environment and that there should be no restrictions on or control of their use.

The ultimate decision as to whether agriculture is contributing to pollution of particular water bodies to such an extent that active control measures are required rests with State or local authorities. To assist these officials in reaching this decision and in choosing appropriate controls, the Federal Water Pollution Control Act Amendments of 1972, Public Law No. 92-500, specify that the Administrator of the Environmental Protection Agency shall, in cooperation with other agencies, provide guidelines for identifying and evaluating the nature and extent of nonpoint sources of pollutants. This two-volume document on control of potential water pollutants from cropland was written by scientists of the U.S. Department of Agriculture in response to this provision of the Act and at the request of the Environmental Protection Agency. Volume I is a User's Manual for guideline development. Here in Volume II we will review some of the basic principles on which control of specific pollutants is founded, provide supplementary information, and present some of the documentation used in Volume I.

Management decisions relating to the control of pollution from cropland involve a careful weighing of potential costs and benefits. Some of the factors affecting these decisions can be visualized by considering the schematic drawing of an agricultural system in Figure 1. The system itself is arbitrary and could consist of a field, a state, or a river basin. Inputs to and outputs from the system can be identified and inputs can be classified as controlled or uncontrolled. Precipitation and solar radiation are uncontrolled and contribute to the stochastic nature of the outputs. The farmer has the ultimate control over the controllable inputs, subject to physical and legal constraints. The outputs can be changed by varying the inputs or the system itself within certain constraints imposed by physical laws.

Most people would agree that the system should be so modified and that the inputs to the system should be controlled at a level that maximizes the net benefits to society attributable to the system. Obviously the modifications and controls chosen depend strongly on the concept of social welfare and must include many costs and benefits not normally accounted for by a land manager.

The costs and benefits of the agricultural use of land depend on the weather and other elements that may be considered as stochastic processes; therefore, the costs and benefits associated with the agricultural use of the land may themselves be considered as stochastic processes. Consider the four sample functions shown in Figure 2. $X_1(t)$ represents the amount of daily precipitation; C_1 (t) represents the amount of a chemical applied to a field and is a stochastic process because the time of application depends on precipitation, stage of crop growth and other factors associated with the particular chemical; $Y_1(t)$ symbolizes daily surface runoff which may transport the chemical to a stream or lake; $Y_2(t)$ represents the amount of chemical transported to surface water; and B(t) represents the benefit process (costs are negative benefits). Social costs include those incurred when surface runoff occurs shortly after a chemical is applied and those due to sediment. Other costs include those normally borne by the farmer. The benefit from the sale of the crop will vary annually as a result of yield variability and the demand of society for the particular crop, expressed as the price. Conceptually the management decision problem is not difficult, but practically it is formidable. First there is the question of uncertainty—we do not know the long-term effects of low, intermittent concentrations of many chemicals on living organisms, including man. Therefore, we cannot estimate the cost attributable to the specific transport of a given chemical. Since this, among many other uncertainties, prevents the selection of control practices and institutional mechanisms that maximize net social benefits, we may wish to state the objective in physical terms. As an example, one could select those control practices which maintained the average chemical concentration below some threshold value for a given percent of the time. If social costs associated with the presence of this chemical in a stream exceeded the benefits attributable to it, this procedure would at least lead to an improved situation. However, as will be shown in subsequent chapters, our technology in predicting the effects of changes in inputs and in the system itself on

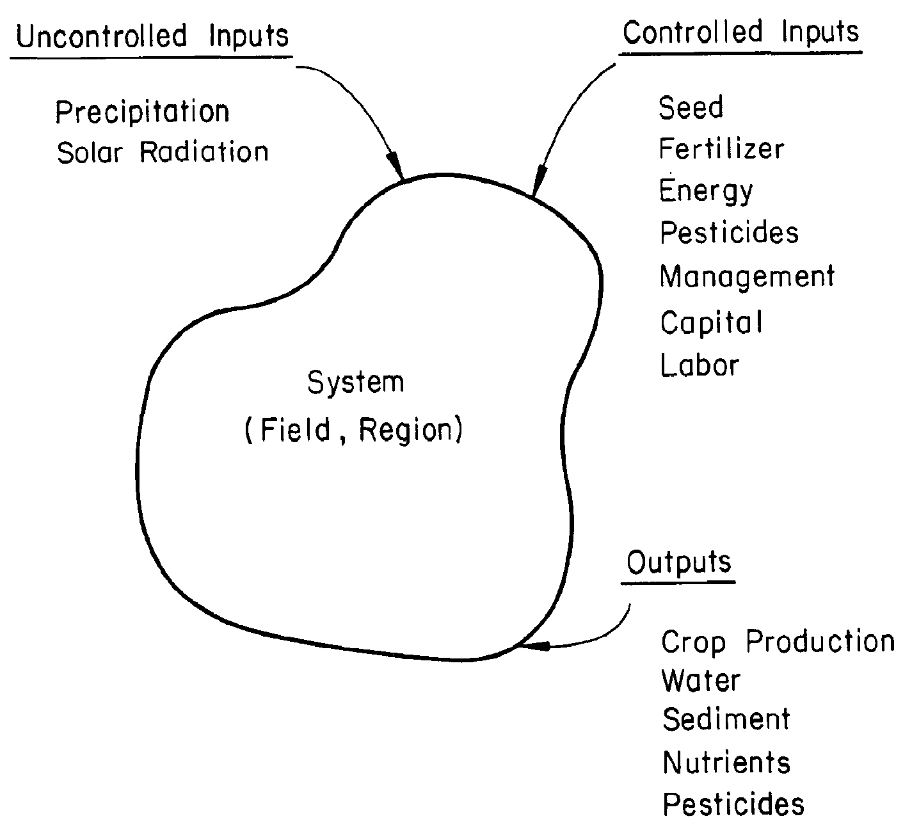

Figure 1.—Agricultural production system.

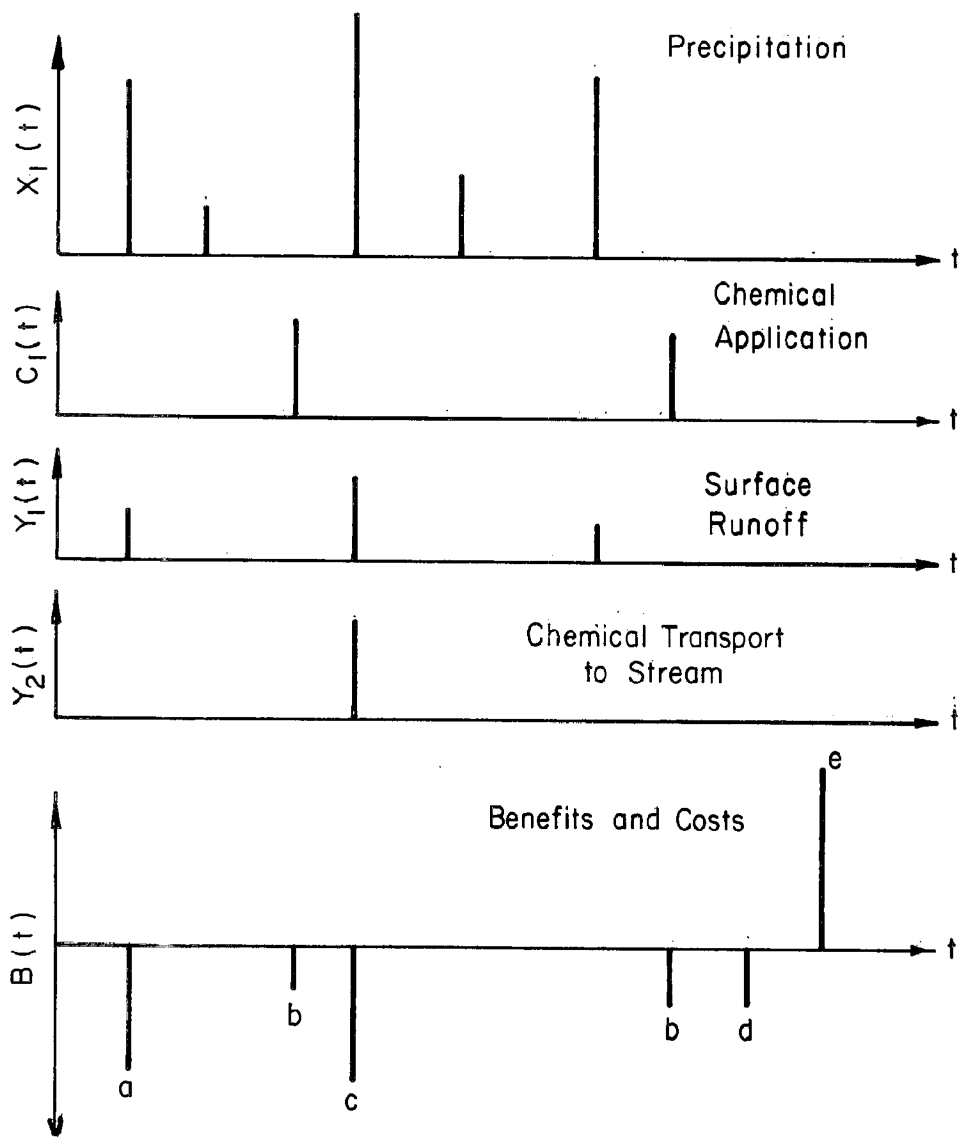

Figure 2.–Sample functions of hydrologic processes and social costs and returns for agricultural system. a: planting cost to farmer; b: cost of applying chemical; c: social cost when chemical is transported to stream; d: harvest cost; e: return from sale of crop.

concentrations of potential pollutants in surface waters is not developed well enough to make this approach feasible. As a last resort, we can use direct runoff, percolation and erosion as surrogate variables with the *assumption* that a change in any of these variables will affect water quality. It must be recognized that reduction of direct runoff or deep percolation may adversely affect water quality in some instances and, therefore, may create water quality and quantity problems for downstream water users who depend on runoff from agricultural lands as a water supply.

Before dealing with specific potential pollutants, it is important to know something about the land resources of the U.S. because this has a significant bearing on the use of agricultural chemicals. *Only the land in the contiguous 48 states will be discussed, since there is so little cropland in Alaska and Hawaii.*

The contiguous 48 states contain 1,899,322,000 acres of land. The nonfederal rural land comprises 1,431,930,000 acres, or 75 percent of the total. The use of this rural land is nearly equally divided between cropland, pasture and range, and forest land (Table 1). A land capability classification system[1] has been developed by the U.S. Department of Agriculture and a summary of the amounts of various classes of soils and their use is given in Figure 3. Class I soils are nearly level, have a low erosion hazard, and are suited to a wide range of plants. They are deep, have high permeability and water-holding capacity, are well drained, and are fairly well supplied with plant nutrients or are highly responsive to fertilizers. Soils in Class II have some limitations that reduce the choice of plants or require moderate conservation practices. They often require special soil-conserving cropping systems, soil conservation practices, water-control devices, or tillage methods when used for cultivated crops. Class II soils usually have gentle slopes and are moderately susceptible to wind and water erosion.

Class III soils are usually found on moderately steep slopes and are more susceptible to water and wind erosion than soils in Class II. They can be used for cultivated crops but require highly effective conservation practices that may be difficult to apply and maintain if erosion is controlled. Class IV soils are also suited for cropland, but they require careful management and are often well suited for only two or three common crops. They are usually found on steep slopes and are highly susceptible to wind and water erosion.

Table 1. Predominant land use for nonfederal rural land in the contiguous 48 states (USDA Statistical Bulletin No. 461)

	Acres
CROPLAND:	
Row crops	160,041,000
Close-grown crops and fallow	132,620,000
Forage crops	77,629,000
Conservation use	39,026,000
Temporary idle	11,235,000
Orchards, vineyards, and bush fruits	5,060,000
Open land formerly cropped	11,592,000
	437,203,000
PASTURE AND RANGE:	
Pastureland	101,061,000
Rangeland	379,929,000
	480,990,000
FOREST LAND:	
Commercial	396,078,000
Noncommercial	62,860,000
	458,938,000
OTHER LAND:	
In farms	27,779,000
Not in farms	27,020,000
	54,799,000

Soils in Classes V, VI, VII, and VIII are limited in their use and are generally considered unsuitable for cultivation. Class V soils have little or no erosion hazard but have other limitations that are impractical to remove. Examples are bottom lands subject to frequent overflow, stony soils, and ponded soils where drainage is unfeasible. Class VI soils are usually limited to pasture, range, forest, or wildlife habitat. However, some Class VI soils can be used for common crops with careful management. Some of the soils are also adapted to special crops such as sodded orchards, blueberries, and similar crops. Class VII soils are not suited for cropland, and Class VIII soils are not only unsuited for cropland, but have limitations so severe that they are restricted primarily to recreation, wildlife habitat, water supply, and esthetic uses.

The erosion hazard of cropland increases sharply from Class I through Class IV soils. Therefore, the larger the cropland acreage on Class III and IV soils, the greater the hazard of erosion. Also, since sediment is a principal transport mechanism for agricultural chemicals, the potential for their loss is much greater on these soils. For example, it is estimated that from one-third to one-half of America's agricultural production depends on fertilizer use. Therefore, if fertilizer use were eliminated, cropland acreage would have to be greatly expanded. Figure 3 shows that any large increase in cropland would have to come from Class III and IV soils. These soils are

[1]National Inventory of Soil and Water Conservation Needs, 1967. U.S. Department of Agriculture Statistical Bulletin No. 461, January 1971.

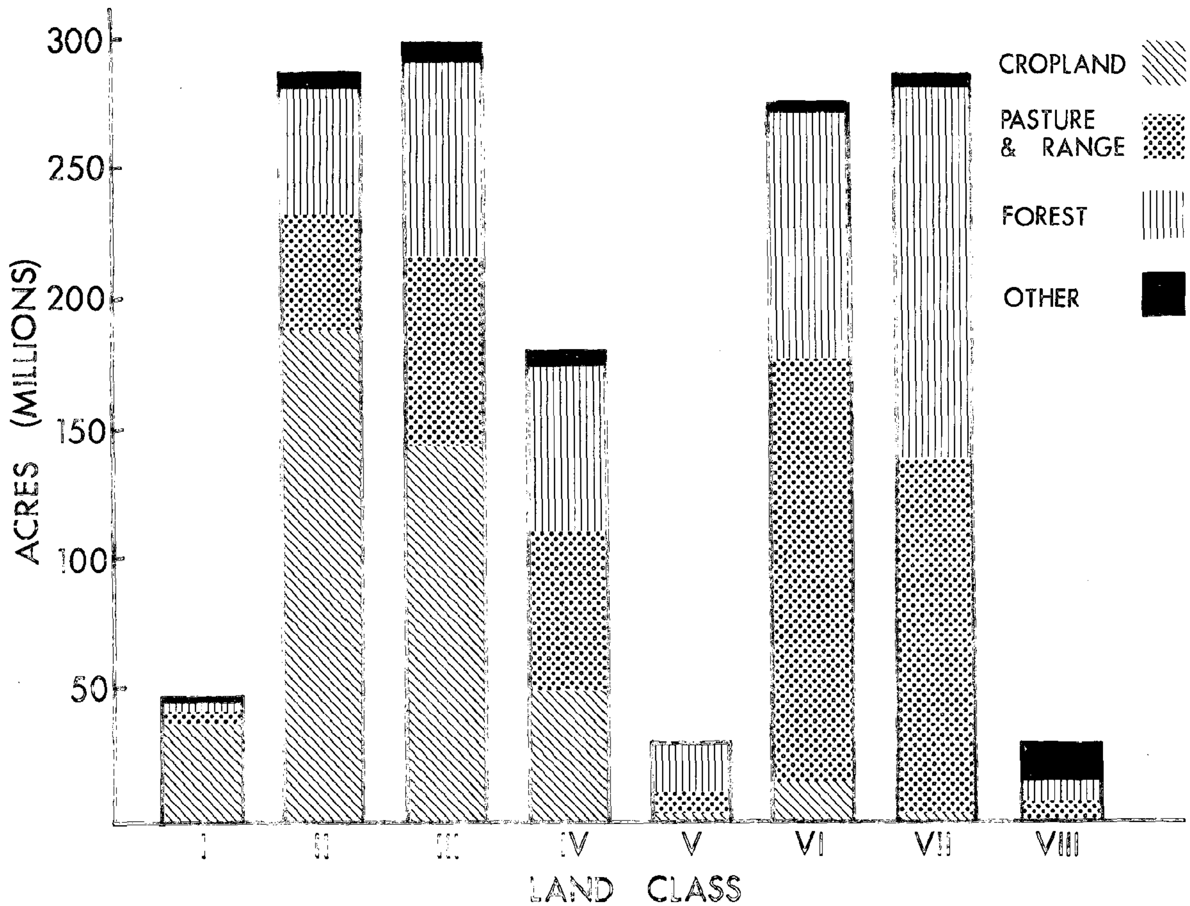

Figure 3.–Use of various classes of land in the 48 contiguous states (based on data from USDA Statistical Bulletin No. 461).

less desirable, not only because they are more erodible, but because they are lower in fertility and yield considerably less than Class I and II soils, particularly when fertilizers are not used.

Unless sediment is controlled at a given level, the loss of agricultural chemicals from equal treatments will usually increase as the soil class number increases. This suggests that one approach to control water pollution from cropland is to concentrate crops to the fullest extent possible on Class I and II soils. These soils are naturally more productive, more responsive to fertilizers because of higher water-holding capacities, and easier to control with respect to sediment losses. In all likelihood, therefore, a high level of food and fiber production with the least impact on the environment would result from using fertilizers and pesticides on the better lands where their effectiveness is high and their loss is small. The use of chemicals on the more erosive soils presents a substantially greater threat to the environment. However, it is possible to use them safely on these soils if a higher level of management is practiced to control sediment and associated chemical losses. The treatments necessary to reduce losses are given in Volume I.

How much agricultural chemicals are affecting the environment is certainly not clear. However, it appears that sediment, nutrient, and pesticide losses can be controlled at an acceptable level by the selection of proper management systems. The challenge, therefore, is to develop appropriate assessment techniques and institutional mechanisms so that controls are used only when needed. Also, recommending control practices for a large area is extremely difficult because the practices are often site-specific. The concepts presented in Volume I and the material presented in the following chapters must, therefore, be considered only as general aids to the decision-making process. Control recommendations for specific sites must be developed by specialists within the area.

CHAPTER 2

HYDROLOGIC ASPECTS OF NONPOINT POLLUTION

D. A. Woolhiser

Water, running over the land surface or percolating through the soil mantle to eventually appear as groundwater runoff, is a potential carrier of pesticides, nutrients and sediment to streams and lakes. Any discussion of nonpoint water pollution from agricultural sources necessarily involves hydrology because water is the primary transport medium.

In this chapter we will consider some hydrologic fundamentals, including basic physical principles and a brief discussion of the stochastic nature of hydrologic processes. An understanding of the stochastic nature of hydrologic processes is important because it affects the interpretation of experimental data. Components of the hydrologic cycle will be described to illustrate the physical basis for modifying surface runoff by agronomic and engineering practices. These components have been aggregated into fairly general mathematical models with the objective of describing agricultural chemical transport. Finally, documentation is provided for most of the 18 direct runoff control practices presented in Section 4.2 of Volume I.

Only those aspects of the hydrologic cycle that are important in nonpoint pollution will be emphasized in this report. Readers interested in a more comprehensive discussion of hydrology are referred to several texts (*23, 32, 64, 115*). Although results of experimental investigations of the effects of land use and treatment on runoff from agricultural lands in the United States were reported as early as 1927 (*84*), only recent experimental work will be considered here because dramatic changes in agricultural practices have introduced time trends in the amount of direct runoff from cropland (*114*).

To understand how nonpoint pollutants move from fields to surface waters, we must first consider the physical form and placement of agricultural chemicals, including nutrients and manures. Then we must consider the various paths they must follow and the conditions (such as temperature, oxygen status, biological activity) they may encounter from field to stream or lake. Form and placement of the potential pollutants are considered in subsequent chapters; in this chapter, we will concentrate on the pathways.

FUNDAMENTALS OF HYDROLOGY

Basic Physical Principles of Hydrology

Two basic physical principles governing the amount and distribution of water on the earth are those of mass conservation and energy conservation. These principles, along with several empirical relationships, form the basis for most mathematical descriptions of hydrologic phenomena.

The principle of mass conservation is frequently illustrated by the hydrologic cycle or by the water budget for an arbitrary volume of soil. Horton's (*52*) qualitative representation of the hydrologic cycle, Figure 1, is useful for introducing some hydrologic terms and expressing the concept that the mass of water on earth is assumed to be constant.

If we consider the sector labeled "surface disposition of precipitation—all forms" in Figure 1, applied to an arbitrary volume of soil with surface area, A, and depth, d, as shown in Figure 2, we can write the conservation equation for some arbitrary period of time, Δt:

$$P + W = Q_s + Q_B + \Delta D + \Delta S + U + E \, , \qquad (1)$$

where:

- P = precipitation received on the area, A
- W = water imported as a result of man's activities
- Q_s = net surface runoff (surface runoff leaving A less surface runoff entering A)
- Q_B = net lateral outflow (may include ground water flow or unsaturated flow)

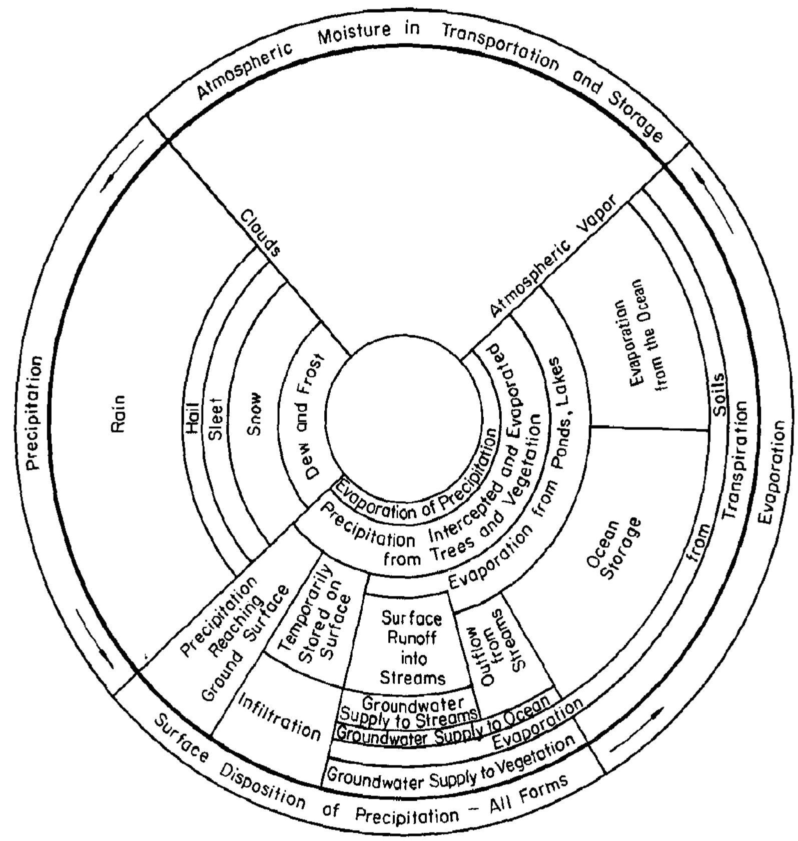

Figure 1.–The hydrologic cycle–a qualitative representation [Horton (52)].

ΔD = increase in surface storage (depression storage and detention storage)
ΔS = increase in soil water storage
U = net vertical outflow through soil or rock
E = evaporation including evaporation from plants (transpiration).

All dimensions are in appropriate depth units. The total water yield for this area, both surface and subsurface, is the difference between the total input of precipitation and imported water, and evaporation, assuming changes in storage are insignificant. Each of these components will be discussed in more detail in the next section.

The amount of evaporation is controlled by the amount of energy available at the layer of soil and air in which plants grow. A conservation of energy equation may be written at this interface, expressing the relation:

Net rate of incoming energy per unit area = net rate of outgoing energy per unit area

$$R_S(1 - \rho) = R_L + G + H + LE \qquad (2)$$

where:

R_S = flux density of total short-wave radiation at the ground surface
ρ = albedo of the ground surface (fraction of incoming short-wave radiation that is reflected)
R_L = net flux density of long-wave radiation
G = heat flux density into the ground
H = sensible heat transfer into the atmosphere
L = latent heat of vaporization of water
E = evaporation rate

Changes in heat storage in the vegetation and the heat used in photosynthesis have been ignored in Equation (2). They would be about 1% of R_S. The terms in Equation (2) are in units of heat energy per unit area per unit time. The magnitude of the terms in Equation (2) may vary substantially. If the soil surface is wet or covered by actively transpiring vegetation, most of the available solar energy may be used to evaporate water. If the soil surface is dry, most of the incoming energy may be used to heat the air.

Equations (1) and (2) are linked by the evaporation term, E. The magnitude of E in Equation (1) is effectively limited by the amount of heat energy delivered to the surface A.

Stochastic Nature of Hydrologic Processes

A set of daily precipitation amounts on a particular field, arranged chronologically, is an example of a time series. Other examples include the daily direct runoff from a field, the daily amount of water percolating below the depth d, or any of the terms in Equations (1) or (2) for an arbitrary period of time. An essential feature of these time series or processes is that they are unpredictable in a deterministic sense. That is, we cannot predict with certainty how much rain will fall tomorrow. These series can be viewed as sample functions of stochastic processes. A stochastic process may be informally defined as a process developing in time in a manner controlled by probabilistic laws (*81*). Many chance mechanisms are important in agriculture. Precipitation is perhaps the most important, but plowing, planting and harvesting dates, and fertilizer and pesticide application dates are certainly not deterministic.

To analyze a time series, one must first assume a mathematical model for the stochastic process which is completely specified except for parameter values that can be estimated on the basis of an observed sample. When the parameter values have been estimated, one can obtain certain probability expressions that may be valuable in decision making. For example, what is the

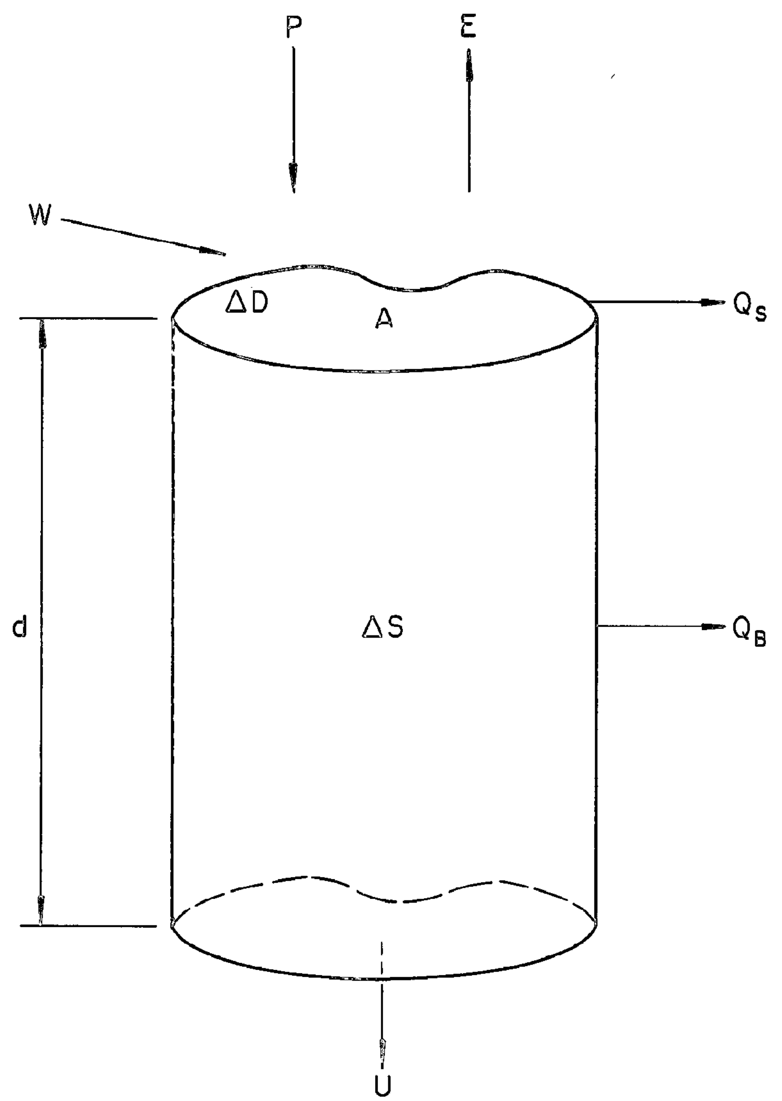

Figure 2.—Control volume for water balance.

probability that the concentration of some substance in runoff from a field will exceed a certain level for 24 hours or more? In an ideal situation, our stochastic models would be constructed in accordance with Equations (1) and (2) so that one might see how a change in land management could affect the probability statement.

We cannot construct such ideal models. However, the stochastic nature of hydrologic phenomena must be appreciated because, if a substance is applied on the land, we cannot guarantee that it will never be transported to a stream. The probability of such transport happening in a particular year may become infinitesimally small, however.

Another important concept is that of stationarity. A stationary process is one where the chance mechanism does not change with time. If we consider the process of surface runoff from a field in continuous corn, we can see that it is not stationary. Not only are there periodic changes within a year caused by seasonal phenomena but also there are long-term trends introduced by changes in agricultural technology such as new tillage implements, new crop varieties and increased fertilizer use. Therefore, one cannot use long time series to estimate parameters because the parameters are changing with time.

COMPONENTS OF THE HYDROLOGIC CYCLE

In this section we will describe individual components of the hydrologic cycle and review some of the mathematical models that have been proposed or used to describe these elements. The discussions will not be comprehensive but will consider those aspects deemed most significant for chemical transport or for reducing surface runoff.

Interception

When rain begins, drops strike plant leaves and stems and are retained on these surfaces by the forces of adhesion and cohesion until a sufficiently thick film of water accumulates that gravitation overcomes these forces. If rain continues, the storage on an individual leaf will become nearly constant, with as much water falling from the leaf as falls upon it. Water will also be lost from the film on vegetation by evaporation. There is some disagreement as to whether this evaporation is a net loss insofar as the water balance of a volume of soil is concerned (*119*). If transpiration is limited by the energy available, evaporation from the water stored on leaves is essentially equal to the amount of water that would be lost by transpiration unless the albedo of wet vegetation is less than that for dry vegetation. If transpiration were limited by soil water content, however, the evaporation from a water film would be greater and part of it could be considered as a net loss. Water evaporated from mulch, dead leaves, stems or trunks could be considered a net loss if energy were not limiting. Rain intercepted by the canopy may subsequently reach the ground by dripping from the leaves or flowing down the stem. If stemflow is significant, it can produce substantial differences in soil-water content over rather small distances (*65*).

Although several have attempted to develop a mathematical description of interception based on physical reasoning (*51, 63*), the models have been rather crude. Many mathematical watershed models do not include an explicit component for interception (*27, 50*). Crawford and Linsley (*26*) combine interception and depression storage into a single lumped storage with depletion by evaporation and transfer to a lower zone storage. Boughton's model (*10*) and the Tennessee Valley Authority model (*110*) assume that precipitation will accumulate in interception storage until a threshold or capacity value is reached. The TVA model uses capacities for forested watersheds of 0.05 inch in winter and 0.25 inch in summer. Saxton et al. (*95*) used a storage amount of 0.10 inch for agricultural crops and showed that evaporation from this source can be several inches per year in a semi-humid climate.

Zinke (*119*) concluded:

"A survey of the data in the literature indicates interception storage amounts for rain of from 0.25 mm to 9.14 mm (0.01 to 0.36 in.) and a similar range for snow, 0.25 mm to 7.62 mm (0.01 to 0.30 in.).

The storages indicate that one would not be greatly in error to estimate about 1.3 mm (0.05 in.) storage capacities for rain for most grasses, shrubs and trees; and 3.8 mm (0.15 in.) for snow for trees."

Jones (*58*) concluded that "a consistent difference in storage capacity for trees, crops, and grass of various heights was not evident."

From this brief review, interception does not appear to have an important influence on runoff or deep percolation from fields. However, an increase in interception is partly responsible for the reduction in runoff caused by conversion from clean-tilled crops to pasture or meadow.

Depression Storage

After interception storage has been filled and the infiltration capacity of the soil is exceeded so that all or part of the soil surface is saturated, water will accumulate in surface depressions. Water stored in depressions either evaporates or infiltrates into the soil--none of it runs off the surface.

Depression storage can be increased by agronomic or engineering practices and, therefore, can be important in reducing direct runoff from fields. For example, under ideal circumstances, as much as 2.5 inches may be stored in contour furrows constructed with a range furrowing machine commonly used in the west (*77*). Level bench terraces with a capacity of over 2 inches have been installed in the deep loess soils of western Iowa (*96*). Doty and Wiersma (*28*) found that the maximum potential depression storage capacity for conventional contouring and for bedding and listing practices ranged from approximately 1 inch for contouring to as much as 3 inches for listing and bedding. The potential surface water storage decreases as land slope increases and is approximately half as great for a 7 percent slope as for a 1 percent slope.

Agronomic and engineering practices to increase depression storage have a transient effect. With annual cropping systems, storage capacity usually is maximum in the planting to first cultivation period, which is frequently the most important for reducing losses of agricultural chemicals by surface runoff. The storage capacity then decreases and reaches a minimum during the harvest to plowing period (*28*). Contour furrows in range and pasture have maximum storage immediately after installation. Erosion and trampling by livestock gradually reduce this storage capacity. For example, contour furrows in eastern Montana had only half their original storage capacity after 6-10 years, and the average effective life (storage $>$ .05 inch) was about 25 years (*77*).

Mathematical descriptions of depression storage usually represent it as a volumetric threshold that must be exceeded before surface runoff occurs (*10, 50*). The Stanford model (*26*) lumps depression storage with interception but does not assume a fixed threshold value. This approach can partially account for the spatial variability of surface detention over the watershed. The parameters in these models are usually found by trial and error or by optimization techniques. Very little information is available that could serve as a guide in choosing values for depression storage based on physical measurements in the field. The work of Doty and Wiersma (*28*) is one exception for fairly simple geometric shapes. Boughton (*10*) suggested that the "random roughness"of soil surface microtopography described by Burwell and others (*15, 16*) might be an adaptable measure of the depression storage.

Although manipulation of depression storage is an obvious method of affecting surface runoff, the amounts of change can be deduced only indirectly by analysis of rainfall and runoff. The curve numbers for contoured and contoured and terraced areas for the Soil Conservation Service method of estimating direct runoff shown in Appendix A reflect some empirical data mixed with judgment. Mathematical models that include depression storage explicitly could be used to predict changes. However, transciency of depression storage and its dependence on precipitation, runoff, and erosion make prediction difficult.

Infiltration

As snow melts or rain falls on the soil surface or drips from the vegetation, the phenomenon of infiltration governs the amount of water that will enter the soil and thereby greatly affects the amount of surface runoff. Some of the physical, chemical and biological characteristics of soil that affect infiltration can be manipulated by man through agronomic and engineering practices. Therefore, changing the infiltration characteristics of soils can profoundly affect the amount of surface runoff as well as the amount of water stored in the soil for plant use.

Characteristics of both the porous medium and the fluid affect infiltration. The porosity, pore-size distribution and tortuosity of soil pores all substantially affect infiltration rates. Sands have higher infiltration rates than silts or clays, which have a higher porosity but much smaller pores. Soil compaction by the trampling of livestock reduces infiltration capacity and increases surface runoff (*85*). From this evidence it can be inferred that compaction by machinery would also decrease infiltration rates and that practices that reduce machine traffic on a field should reduce surface runoff.

Raindrop impact on bare soil breaks up soil aggregates into their component particles or much smaller aggregates. These particles or small aggregates can be carried into larger pores by water and form a thin surface layer that has low hydraulic conductivity. This surface layer may then control the infiltration rate (*33, 47*). Vegetation or mulches protect the soil surface from raindrop impact and can prevent crust formation.

Dense vegetation with massive root systems and farming systems that leave substantial amounts of plant residues near the surface maintain high soil organic matter content and promote aggregate stability, thus maintaining high infiltration rates. Vegetation also has a higher evapotranspiration rate between rains than evaporation from bare soil, thus the soil water content is reduced at the beginning of the next rain which increases the rate and amount of infiltration.

Tillage can increase the volume of large pores near the soil surface and thereby increase infiltration rates. The effect is transient, however.

Frozen soil usually has a lower infiltration rate than unfrozen soil. If the soil is frozen while wet, a dense, nearly impermeable mass may result. However, if frozen while dry, some soils will show little change in infiltration rate (*76*). The effect of increased viscosity of the water is apparently compensated for by the structural change caused by freezing. Frost usually penetrates deeper if the soil is bare than if it is snow covered. Therefore, practices that prevent snow from blowing away tend to lessen frost penetration but the additional snow deposited may increase runoff.

Modern infiltration theory based on the theory of unsaturated flow or two-phase flow in porous media has provided a basis for understanding infiltration behavior. This theory has been presented in several recent texts or reviews dealing with theoretical aspects of infiltration (*8, 38, 46, 75, 83*).

Although infiltration theory is useful in explaining observed infiltration phenomena, it has just begun to be used in quantitatively estimating the effects of agronomic or engineering practices on infiltration and surface runoff. The partial differential equations describing infiltration must be solved by numerical methods—a time consuming and costly task if one wishes to find long-term average effects or distribution functions of surface runoff. Also, this approach, with its strong physical basis, requires costly and difficult measurements of soil conductivity and diffusivity (*109*).

Because of these difficulties, several infiltration equations, either entirely empirical or based on simplifications of the more general formulations, have been used. Equations presented by Horton (*53*) and Holtan (*48*) are examples of the former. Green and Ampt (*39*), Philip (*82, 83*), Smith (*104*), Mein and Larsen (*69*), and Brustkern and Morel-Seytoux (*14*) used either simplifications of the basic equations or algebraic approximations of numerical solutions of the basic equations. The first three of these apply only to infiltration from a ponded surface rather than to rainfall conditions. Although solution of these equations is simpler and less costly than solution of the more rigorous partial differential equations, the question of parameter estimation remains. Usually they are estimated for different soil and cover conditions from infiltrometer experiments on small plots or data from small watersheds. Musgrave and Holtan (*76*) reviewed much of the data available before 1964. Holtan and his associates attempted to develop techniques for estimating parameters in the Holtan equation by using information available in soil surveys (*34*) or by estimating parameters for various land-use or cover factors (*49*).

Of the hydrologic models considered, none includes an infiltration component based on the numerical solution of unsaturated flow or two-phase flow in porous media. The USDAHL model (*50*) utilizes Holtan's equation. The Boughton model, the TVA model and the Stanford model utilize empirical lumped storage infiltration components, although the Stanford model attempts to account for spatial variability by assuming an invariant statistical distribution of infiltration capacity. The USGS model (*27*) utilizes an adaptation of the Green and Ampt equation.

Soil Water and Groundwater

Water stored in the soil and rock is frequently separated into two components: the saturated or groundwater zone and the unsaturated zone between the groundwater and the surface. Water moves within the unsaturated zone in response to gravitational and capillary potential gradients. It may move generally downward during rainfall or snowmelt and generally upward after a long, dry period, or it may move upward near the surface and downward in the lower part of the profile simultaneously. In general, water movement in the unsaturated zone will be predominantly vertical.

In some soils, a rather permeable topsoil is underlain by a slowly permeable clay layer. If infiltration is rapid enough, the surface soil may become saturated, resulting in flow which is predominantly in a lateral downslope direction and is known as interflow. This water may reappear on the surface some distance downslope or at the foot of the slope. Hydrologists generally agree that a flow mechanism such as interflow exists; however, there is some argument about its importance. Dunne (*31*) concluded from his measurements of subsurface storm flow in Vermont that interflow (subsurface storm flow in his terminology) did not contribute significantly to flood hydrographs. This does not mean, however, that interflow is unimportant to water quality in some regions. Minshall and Jamison (*71*) presented data suggesting that interflow can exist on Midwest claypan soils.

An interflow runoff component is included in the Stanford model, the TVA model and the USDAHL-70 model. However, the volume of interflow runoff has not been compared with field measurements because of the difficulty in making such measurements. Therefore, it is difficult to ascertain if computed volumes are realistic or are merely a result of curve-fitting procedures.

As Amerman (*3*) has pointed out, the separation between the saturated zone and the unsaturated zone is unnecessary from the physical point of view and is possibly misleading. Figure 3(a) shows a hypothetical transverse cross section through a valley during a relatively dry period and Figure 3(b) shows a similar section during a wet period. Under steady-state conditions, the streamlines would represent the path lines of water molecules or dissolved materials. However, hydrologic systems are usually unsteady so the streamlines are continually shifting. The medium shown in this sketch is isotropic so the streamlines are perpendicular to the equipotential lines. In an anisotropic porous medium that contained a relatively impervious layer, for example, this would not be true.

Figure 3 illustrates some important points about transport of dissolved chemicals. Suppose that in Figure 3(b) the soil surface from point A to point B was within a single field. If we assume a steady state, the path line from A to the stream is much shorter than that from B to the stream. Therefore, a soluble chemical that leached below the root zone on a particular day would take

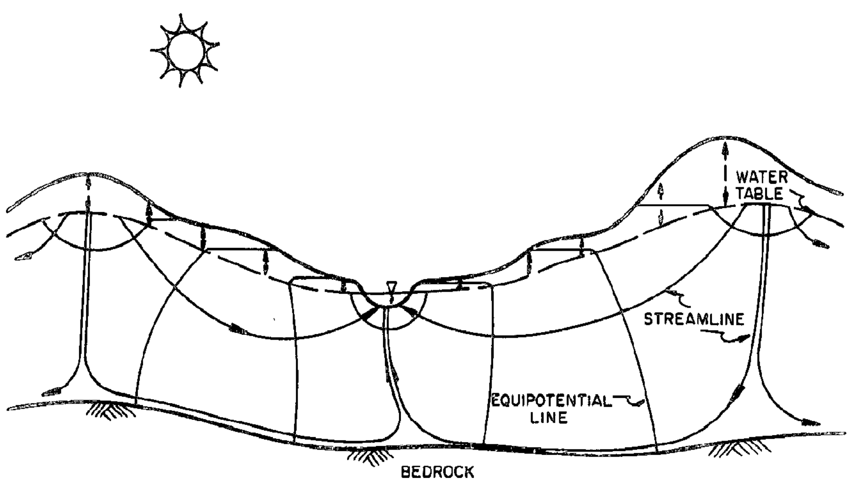

Figure 3(a).—Cross section of hypothetical hydrological system during a relatively dry period.

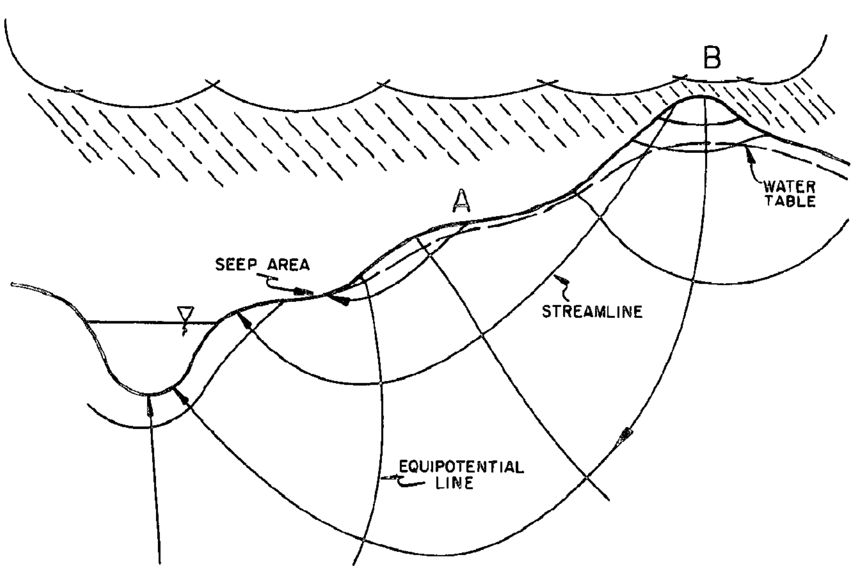

Figure 3(b).—Section of hypothetical hydrologic system during a wet period. [From Amerman, (3)] much longer to reach the stream from point B than from point A. Hydrodynamic dispersion will also affect the arrival time at a stream but has a relatively small effect compared with that of macroscopic flow (*78*). How much delay time might be involved between the arrival times of chemical constituents at the stream? This time, of course, depends on the path length and velocity. Some numerical models can answer this question if the system geometry and hydraulic characteristics of the medium are known (*12, 86, 100*). As a crude approximation we might use the results of Carlston (*20*), who found that the mean residence time of groundwater recharge in a Wisconsin drainage basin was about 45 days. If we assume that the time of travel for a particle following the streamline originating between A and B is equal to the mean residence time and that the path length of A is half the mean and the path length of B is 1.5 times the mean, the arrival of a slug of chemical distributed uniformly over the field extending from A to B might appear at the stream over an interval of 45 days.

Of course, the physical, chemical and biological processes that affect the particular constituent during its travels through the porous medium must also be considered. For example, if we are concerned with nitrate transport, some zones along the flow path may be anaerobic and contain carbon. Under these circumstances, bacteria may convert the nitrate to harmless N gas. Such conditions might well exist in the seep area shown in Figure 3(b).

Legrand (*62*) discussed the patterns of contaminated zones of water in the ground. Although he considered contamination sources of small areal extent (point sources), his concepts can be readily applied to nonpoint sources. He noted that when contaminants move through the unsaturated zone and reach the water table, "enclaves" of contaminated water extend from the source in the direction of groundwater movement, as shown in Figure 4. If the contaminant is not adsorbed or chemically or biologically transformed, the enclave will terminate at a stream (Field A, Fig. 4) and may cause pollution. Because of additional water entering the stream, the contaminant may be diluted to a harmless level at a point C downstream.

The boundary of the enclave shown in Figure 4 assumes a constant inflow of the chemical uniformly distributed over the field. As a rule, inputs will be intermittent; therefore, the pattern may consist of a series of smaller enclaves moving toward the stream completely surrounded by uncontaminated water. The dashed line emanating from the lower boundary of field A terminates before reaching the stream, illustrating the situation in which some of the chemical may pass through a zone where chemical or biological reactions may reduce its concentration to harmless levels before it reaches a stream. The same situation holds for the contaminant moving from field B. Enclaves will change in areal extent and in shape as the water table changes its configuration naturally or by pumping of wells.

Robbins and Kriz (*92*) presented a comprehensive review of groundwater pollution caused by point and nonpoint agricultural sources. Their concern was primarily with measurements of water quality within the enclaves of groundwater contamination, not with the effects on water quality in streams and lakes. For an excellent review of mathematical models describing movement of chemicals in soils, see Boast (*9*).

The subsurface transport of agricultural chemicals from a field to water bodies is obviously very complicated. Although we have a qualitative understanding of such transport, much uncertainty is involved in predicting when and how much of a chemical may reach a stream or lake, or how much the amount can be reduced by control practices.

We can, however, identify certain goals of subsurface water management on agricultural land. Maintaining adequate water in the root zone and encouraging a vigorous crop are advantageous from both the crop production and water quality standpoints. Deep percolation will occur in most humid and sub-humid climates. Variations in soil characteristics will lead to substantial differences in annual percolation, as shown in Figure 12, Vol. I, and in Appendix B of this volume. In those areas with substantial deep percolation, soluble agricultural chemicals must be applied with more care.

Evapotranspiration

The sum of evaporation from the soil surface and transpiration from plants is called "evapotranspiration" and represents the transport of water from the earth to the atmosphere. It is important in agriculture because it is required for crop growth. It is important in the loss of potential pollutants from cropland because it affects the volume of direct runoff and the amount of soil water that percolates to the saturated zone. Evapotranspiration is obviously a major component in the hydrologic cycle--it transports about 70 percent of the water that falls on the conterminous United States back to the atmosphere. This percentage can vary from 100 in arid regions to about 50 or less in some mountainous areas of the U. S.

Three physical requirements must be met for evaporation from a surface to continue: 1) There must be a supply of heat to convert liquid water to vapor, 2) the

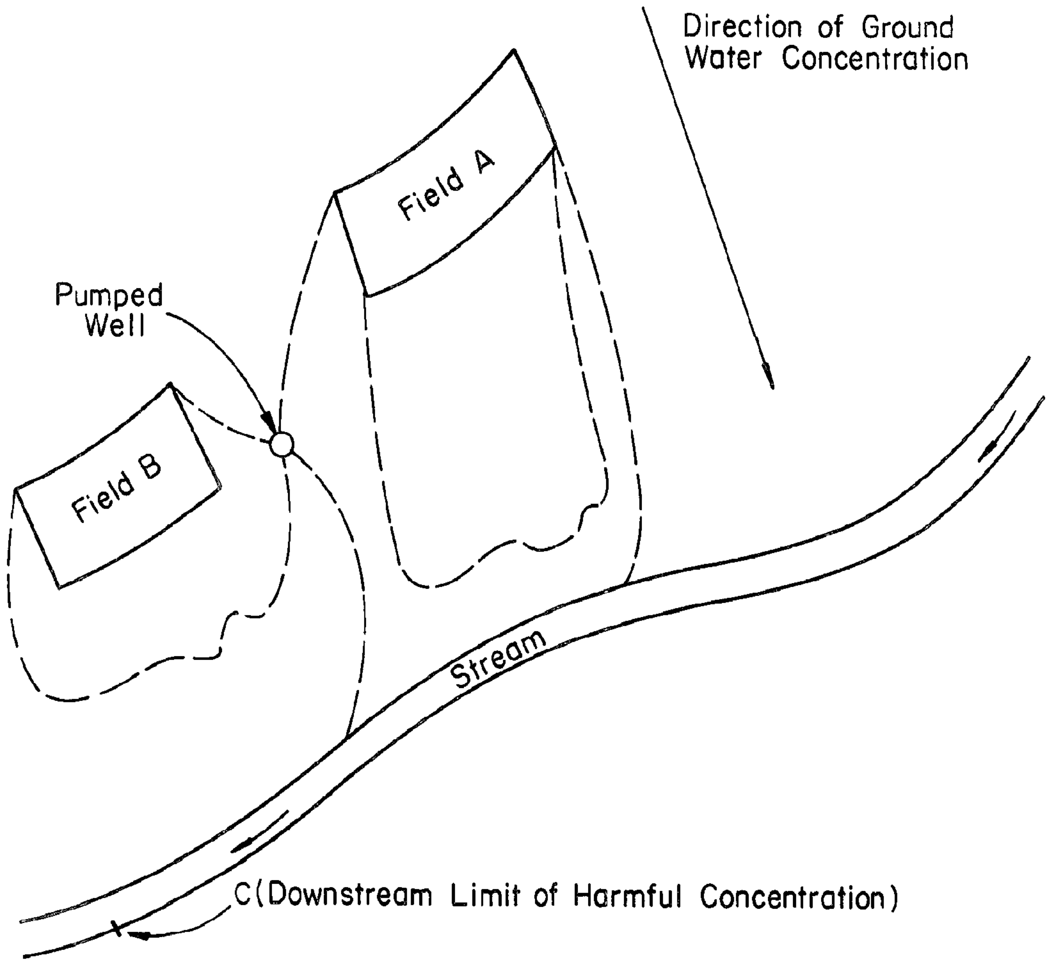

Figure 4.—Plan view of water-table aquifer showing enclaves of groundwater with high concentrations of a soluble material added to fields A and B.

vapor pressure of the air must be less than that of the evaporating surface, and 3) water must be continually available. Evapotranspiration, through the latent heat term, is a component of the energy balance, Equation (2), as well as the hydrologic water balance, Equation (1).

When water is not limiting, the evapotranspiration rate is limited by the radiant energy and advected energy available. Therefore, a lower limit exists for total runoff—the difference between precipitation and potential evapotranspiration. Potential evapotranspiration is defined as the hypothetical rate of water loss from a large, homogeneous area of continuous green crop, under the given meteorological conditions, when there is no resistance to water supply at the evaporating surface (*112*).

At most locations in the United States, soil water is limiting some time during the year, so the actual evapotranspiration will be less than the potential even if the ground is fully covered by a crop. With annual row crops, the ground will not be covered by a transpiring crop canopy for a substantial period of time, so evapotranspiration will be less than from grasses and the total runoff will be greater. This is one reason why conversion from row crops to meadow or pasture usually reduces runoff.

The physics of evaporation and evapotranspiration is discussed in several texts (*19, 93, 108*). Here, we will briefly outline the approaches that have been used to estimate actual evapotranspiration from cropland. In general, the models used consist of a continuity relationship, a means of computing potential evapotranspiration,

E_0, which serves as the upper limit of the actual evapotranspiration rate, a method of computing actual evapotranspiration, E, as a function of soil water content when it is below some critical level, and a means to modify E if the ground is not fully covered by a crop canopy.

Potential evapotranspiration can be computed by the energy budget method, the aerodynamic method, or a combination of the two (*113*). It can also be estimated empirically from evaporation pan data. The equation frequently used is:

$$E_0 = K\, E_p \ ,$$

where K is a "pan coefficient" and E_p is the evaporation from a standard pan. Saxton et al. (*94*) found that daily evapotranspiration computed by adjusted pan evaporation was highly correlated ($R^2 = 0.87$) with E_0 calculated by the combination method.

It now seems to be accepted that actual evapotranspiration can be less than potential at soil water contents above the wilting point. Baier (*4*) presented a comprehensive review of this subject. The procedure used in the simulations of potential percolation in Volume I and documented in Appendix B of this volume uses a relationship between E/E_0 and available water that is similar to those presented in the literature. When evapotranspiration estimates are needed for different stages of crop development, the evaporation rate can be corrected by using a crop coefficient, K_p, that varies according to the stage of growth of the crop (*57*), or the ratio E/E_0 may be related to the leaf area index, LAI (*41, 90*). The methods used for the simulations presented in Volume I are described in more detail in Appendix B. For the extensive simulation study of percolation and nitrate leaching in Volume I, a physically more realistic but more complex model such as presented by Richardson and Ritchie (*89*) or Saxton et al. (*95*) would have been difficult to use with existing time constraints.

Surface Runoff

As the transport medium for dissolved chemicals and for sediments with their adsorbed chemicals, surface runoff is an important link between fields and streams or lakes.

Surface runoff begins when the rainfall (or snowmelt) rate exceeds the infiltration rate of the soil and depression storage is filled. Surface runoff is classified somewhat arbitrarily as either overland flow or channel flow. Overland flow is sometimes considered to be thin sheet flow over a relatively smooth surface. However, a more general and realistic definition would be the flow that is outside of the well-defined channel system. The mean velocity of overland flow is directly related to the slope (laminar flow) or the square root of the slope (turbulent flow) and is inversely related to the hydraulic resistance of the surface. The hydraulic resistance varies widely, depending on the surface characteristics, from a Mannings resistance coefficient of 0.02 for bare soil to 0.4 for a dense turf (*118*). Such differences in hydraulic resistance would result in water being about six times deeper on the turf than on the bare soil for the same discharge. The velocity, of course, would be only one-sixth of that on the bare soil. The greater depth on the dense sod would allow much more time for infiltration after the rainfall stopped, resulting in less runoff even if the infiltration characteristics of the soils were the same. The decreased shear stress on the soil with a sod cover would also result in a much lower erosion potential.

Bailey, Swank and Nicholson (*5*) described the modes of pesticide transport into and within the moving liquid boundary during rainfall. The same processes would also apply to nutrient transport by surface runoff. This transport process consists of four mechanisms, as shown in Figure 5: 1) diffusion and turbulent transport of the dissolved chemical from the soil water into the overland flow film, 2) desorption of the chemical from soil particles into or toward the moving film, 3) dissolution of stationary particulate matter trapped at the boundary, and 4) scouring of particulate matter and its subsequent dissolution.

From a consideration of these processes, one could infer that practices that reduce runoff velocities and prevent scour of particulate matter might reduce chemical transport even if the total volume of runoff were not reduced. However, the most effective practices would be those that increased infiltration rates so that more chemicals could be carried into the soil by bulk-flow transport. An increase in depression storage would have a similar effect.

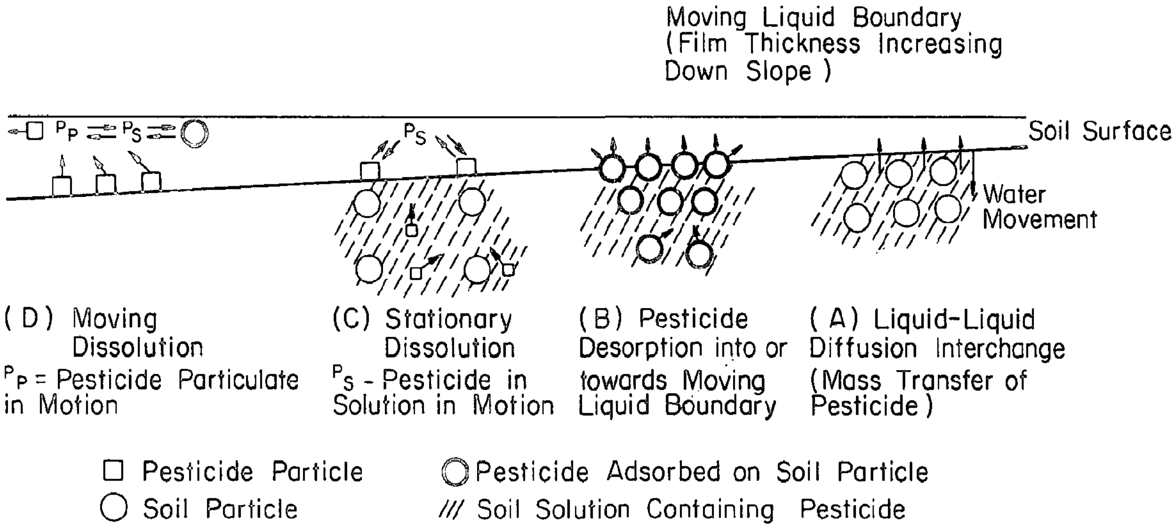

Figure 5.–Modes of pesticide transport into and within the moving liquid boundary during a rainfall event. [From Bailey, Swank and Nicholson (5)]

AGRICULTURAL CHEMICAL AND SEDIMENT TRANSPORT MODELS

Models of agricultural chemical and sediment transport (which may be interpreted to include predictions made by them) represent our descriptions of how water, sediment, and chemicals move on fields or watersheds under existing or proposed conditions. The models should not violate the basic physical principles of hydrology and should incorporate principles of chemistry and biochemistry needed to describe chemical behavior in a biological system. They will also include a number of empirical relationships.

Comprehensive models of the transport of water (hydrologic models) have been used for about 10 years. Several of them have been discussed in previous sections of this paper. Although special purpose water quality models were developed as early as 1925 (*107*), general transport models were not developed until 1967 (*54, 55*).

Development of agricultural chemical transport models started around 1970. There have been several reviews of the "state of the art" and the philosophy of modeling chemical transport (*1, 24, 37, 59, 61, 116, 117*).

Bailey et al. (*5*) have developed a conceptual model of pesticide runoff from agricultural lands, and several quite general models are in the development and testing stage (*13, 25, 37*).

The model developers usually started with a hydrologic model that was developed for some other purpose and added components for chemical transport. The structure of the hydrologic model used thus served as a constraint on the transport model, imposing all of its constraints and shortcomings. When these models have been tested more thoroughly, many of the shortcomings may be shown to be in the structure of the hydrologic model.

These detailed models may be quite useful in an intensive study of a particular field or watershed, but they are far too complex for the extensive scope of this report. Field data available for calibrating or testing these models is also limited. Because of the present lack of knowledge in modeling movement of agricultural chemicals, we used potential direct runoff and potential percolation as surrogate variables in Volume I. We implicitly assumed that if surface runoff were reduced, the transport of chemicals would also be reduced. The models used to estimate potential direct runoff and potential percolation are described in Appendices A and B, respectively.

AGRICULTURAL PRACTICES TO CONTROL DIRECT RUNOFF

Eighteen practices for controlling direct runoff, designated as R1 though R18, are presented in Volume I (Table 14 and Section 4.2). Practices that reduce erosion will usually reduce direct runoff, although to a lesser extent. Therefore, the first 16 runoff control measures have been assigned the same reference numbers as the identical erosion control measures; only the alphabetical prefixes differ. These agronomic or engineering practices constitute means whereby direct runoff may be reduced as compared to direct runoff from an index crop—summer row crop (corn) with straight rows. These practices are discussed in Volume I without supporting documentation. Table 1 contains citations of articles supporting statements made in Volume I and of articles containing closely related information that may be helpful in evaluating individual practices.

Most of the research cited in Table 1 was completed within the last 15 years. Earlier work is not cited because of possible nonstationarity caused by changes in agricultural practices. The percentage reductions in runoff are shown without any indication of statistical significance. However, the decreases reported are consistent with the physical basis of hydrology discussed in previous sections. Ranges in response for individual practices are usually attributable to soil and climatic differences and to sampling variability.

Additional documentation of land use and treatment is given in the reports of the Cooperative Water Yield Procedures Study Project (*101, 102*) and in several recent reviews (*11, 61, 74*).

Table 1. Bibliography on practices to control direct runoff (Volume I, Section 4.2)

Runoff Control Practice			Citations	Significant Subjects
No.	Page No. in Vol. I.	Description		
R1 •	71	No-till Plant in Residues of Previous Crop	Harrold, Triplett and Youker (42)	Comparison of runoff and soil loss from no-till and conventional tillage corn at Coshocton, Ohio.
			Harrold and Edwards (45)	Single-storm runoff from a no-till field of corn on a 21% slope was less than that from straight-row corn field on a 6.6% slope but slightly greater than that from contoured corn on the 6.6% slope.
			Harrold, Triplett and Youker (43)	Comparison of 3 years of runoff and soil loss data from no-till and conventional tillage corn at Coshocton, Ohio.
			Harrold, Triplett and Youker (44)	Five-year average May-Sept. runoff was 0.44 inch for conventional and 0.04 inch for no-till corn at Coshocton, Ohio.
			Smith and Whitaker (103)	In a 3-year period at McCredie, Mo., runoff from corn with conventional tillage averaged 5 in; runoff from no-till fields was 6.7 in.
R2	71	Conservation Tillage	Allis (2)	Over a 9-year period direct runoff from a subtilled field in a corn-oats-wheat rotation was 19% less than from straight-row fields in the same rotation.
			Free and Bay (36)	Runoff from a field of corn with mulch tillage was greater than runoff from conventional tillage.
			Mannering and Burwell (66)	Review runoff and erosion data from various mulch tillage practices.

Table 1. (continued)

Runoff Control Practice			Citations	Significant Subjects
No.	Page No. in Vol. I.	Description		
			Moldenhauer *et al* (72)	Runoff from a till-plant field was slightly less than from a conventionally-tilled field for rainfall applied with a rainfall simulator in early June.
			Onstad (79)	Till-plant tillage up and down the slopes reduced runoff 42% over a 6-year period as compared to conventional tillage.
R3	72	Sod-Based Rotations	Jamison, Smith and Thornton (56)	Review of experiments at McCredie, Mo..
			Epstein and Grant (35)	Comparison of runoff and erosion from continuous potatoes and potatoes-sod-oats rotation at Presque Isle, Me.
			Barnett (7)	Rotation studies at Watkinsville, Ga.
			Burwell and Holt (17)	Compares runoff from corn-oats-hay rotation with runoff from continuous corn in west-central Minnesota.
			Carter, Doty and Carroll (22)	Runoff from Bermudagrass-corn rotation compared with continuous corn at Holly Springs, Miss.
			Mannering, Meyer and Johnson (67)	Evaluated effect of sod-based rotation on soil loss and infiltration using a rainfall simulator.
			Moldenhauer, Wischmeier and Parker (73)	Runoff measured from corn-oats-meadow rotation and from continuous corn.
			Saxton and Whitaker (98)	Runoff measured from corn, small grain, meadow rotation and from continuous row crops.
			Soil Conservation Service (105)	Runoff curve numbers established for rotation meadow and row crops in rotation.
R4	72	Meadowless Rotations	Jamison, Smith and Thornton (56)	Review of crop rotation experiments at McCredie, Mo.
			Richardson (88)	Measurements of runoff from cotton, corn, oats rotation and oats, clover, cotton and grain sorghum rotation at Riesel, Tex.
R5	73	Winter Cover Crop	Mannering and Burwell (66)	No runoff reduction for corn interseeded with legumes at LaCrosse, Wis.
			Smith and Whitaker (103)	A small grain cover crop planted after corn was removed for silage reduced runoff substantially (5.5 in. vs. 11.5 in.)
R6	73	Improved Soil Fertility	Jamison, Smith and Thornton (56)	Average annual runoff from plots at McCredie Mo. ranged from 6.8 to 11.7 inches during 1941-50. Lowest runoff was for well fertilized pasture; highest runoff was for unfertilized corn-oats rotation.

Table 1. (continued)

	Runoff Control Practice		Citations	Significant Subjects
No.	Page No. in Vol. I.	Description		
			Carter, Dendy and Doty (21)	Average runoff was 9.01 inches from improved fertilized pastures and 17.9 from unfertilized pastures for a 6-year period. Experiment was at Holly Springs, Miss.
			Moldenhauer, Wischmeier and Parker (73)	For a 10-year period at Clarinda, Iowa, runoff from continuous corn receiving nitrogen fertilizer was 25% less than runoff from unfertilized corn.
			Saxton and Whitaker (98)	Average annual runoff from row crops with fall fertilization was 1.10 inches as compared with 2.16 inches from row crops receiving only starter fertilizer.
			Wischmeier (114)	The ratio of runoff from corn land to runoff from adjacent fallow decreases with increases in corn yield. Much of the increase in corn yield is caused by higher fertilizer use.
R7	73	Timing of Field Operations	None	
R8	73	Plow-Plant Systems	Free and Bay (36)	Average growing season runoff for plow-plant corn was less than runoff from conventional corn at Marcellus, N.Y.
			Mannering and Burwell (66)	Runoff from simulated rainfall on plow-plant corn was less than that from conventional corn.
			Wischmeier (114)	Reports three studies that show reduced runoff for plow-plant corn.
R9	73	Contouring	Harrold and Edwards (45)	Runoff from a 5-inch rain was 2.30 inches from a contoured corn field and 4.40 inches from a conventional straight-row field.
			Allis (2)	Over a 9-year period direct runoff from a contoured field in a corn-oats-wheat rotation was 32% less than from straight-row fields in the same rotation.
			Carter, Doty and Carroll (22)	Example of 45% runoff reduction by contouring in northern Mississippi.
			Onstad (79)	Contouring in addition to till-planting is more effective in reducing runoff than till plant alone.
			Wischmeier (114)	Refers to three reports showing runoff reduction by contouring.
			Ritter *et al* (91)	Demonstrates that pesticide losses were greater from contoured fields than from ridged fields (R-14).
			Burwell *et al* (18)	Examples of comparative nutrient losses from terraced and contoured fields.

Table 1. (continued)

	Runoff Control Practice			
No.	Page No. in Vol. I.	Description	Citations	Significant Subjects
			Onstad and Olson (80)	Runoff from contoured and conservation tillage fields in corn.
			Spomer, Heinemann and Piest (106)	Compares runoff from contoured corn with level terraced corn and with meadow.
			Soil Conservation Service (105)	Runoff curve numbers established for contoured row crops.
R10	73	Graded Rows	Moldenhauer *et al* (72)	Graded rows on slopes of 3.4 to 9% did not reduce runoff.
R11	73	Contour Strip Cropping	None	Effects inferred from runoff reduction by meadow.
R12	73	Terraces	Baird and Richardson (6)	Terracing alone on heavy clay soils of Texas Blacklands had little effect on runoff volume.
			Richardson (88)	Effects of conservation practices including terracing on runoff.
			Spomer, Heinemann and Piest (106)	Level terraces drastically reduced surface runoff in western Iowa but groundwater flow increased.
			Burwell *et al* (18)	Level terraces reduced discharge of water, sediment, nitrogen and phosphorus.
			Saxton and Spomer (96)	Fourteen percent of water yield from level terraced watershed was surface runoff. Sixty-four percent of water yield from contour watershed was surface runoff.
			Saxton, Spomer and Kramer (97)	Effects of level terracing on runoff and erosion.
			Soil Conservation Service (105)	Runoff curve numbers established for graded terraces.
R13	74	Grassed Outlets	None	No data available on effects of grassed outlets on surface runoff.
R14	74	Ridge Planting	Mannering and Burwell (66)	Ridge planting on contour reduced direct runoff.
			Moldenhauer *et al* (72)	Ridge planting on graded rows did not reduce direct runoff from a simulated rain.
			Ritter *et al* (91)	Ridge planting reduced pesticide runoff.
R15	74	Contour Listing	Mannering and Burwell (66)	Cite Iowa study where annual direct runoff from contour-listed corn was 55% less than that from straight-row planting up and down the slope.
R16	76	Change in Land Use	Jamison, Smith and Thornton (56)	Direct runoff from pasture and meadow was lower than that from corn in central Missouri.

Table 1. (continued)

No.	Runoff Control Practice: Page No. in Vol. I.	Runoff Control Practice: Description	Citations	Significant Subjects
			Dragoun (29)	At Hastings, Nebr. average annual direct runoff was 0.20 inch from watershed in grass and 5.24 inches from cultivated fields in row crops.
			McGuinness and Harrold (68)	Water yield decreased when watershed was reforested.
			Rice and Dragoun (87)	Reseeding cropland with perennial prairie grasses reduced runoff by 94% in a 2-year period.
			Saxton and Whitaker (98)	Comparison of direct runoff from pasture and meadow.
			Spomer, Heinemann and Piest (106)	Comparison of direct runoff from perennial grass, contoured corn and level-terraced corn.
			Thomas, Carter, and Carreker (111)	Bermudagrass meadow reduced runoff.
			Wischmeier (114)	For nearly 5000 plot-years of data analyzed, runoff from row crops averaged 12% of total rainfall, while that from meadow averaged 7%.
			Hanson *et al* (40)	Effects of grazing intensity on direct runoff from rangeland.
			Soil Conservation Service (105)	Runoff curve numbers established for pasture and meadow.
R17	76	Other Practices	Dragoun and Kuhlman (30)	Contour furrowing reduced runoff from pastures.
			Mickelson (70)	Storing runoff in leveled areas for crop production.
			Neff (77)	Storage capacity of contour furrows in rangeland.
			Schwab and Fouss (99)	Surface runoff and tile flow from fields with corn and grass cover.
R18	76	Construction of Ponds	Langbein, Hains and Culler (60)	Hydrology of ponds and stock-water reservoirs.

RESEARCH NEEDS

Research on the effect of land management practices on hydrology has usually involved three steps: 1) intensive experimental measurements on plots and watersheds, 2) analysis of the data using some type of a mathematical model, and 3) generalization of results for more extensive application.

Measurements made in the first step are frequently governed by the model that is to be used in the second step. For example, in most of the experimental work examined, only rainfall and runoff were measured. This was adequate when the only question was "Will treatment A reduce surface runoff?" and when the study could be maintained for enough time to obtain statistically significant results. Unfortunately, we can no longer afford this luxury of time. Policy decisions must often be made quickly and by the time we have statistical significance, the practice may be obsolete.

The alternative is to develop more detailed models to use as a framework in analyzing the data and to obtain more intensive measurements in a shorter time. The experimental data can be used to estimate model parameters and techniques must be developed for predicting parameters from readily obtainable physical measurements. Simulation can then be used to evaluate the stochastic properties of the system and to examine long-term effects.

Stochastic models of point and areal precipitation must be developed and the parameters regionalized by mapping or other techniques. As plant growth models and other biological processes are included in hydrologic models, the stochastic inputs must be expanded to include temperature and radiation. Obviously, the joint probability structure of precipitation, radiation and temperature must be maintained.

Prediction of runoff from complex areas is still difficult and needs a great deal of work from the standpoint of water quality. If concentrations of the chemical are important we must estimate the joint probability structure of discharge of chemicals to a stream and the quantity of water in the stream.

A second generation of agricultural chemical transport models should be developed after the first generation models have been tested and their strengths and weaknesses identified. Material models, systems which retain many of the important characteristics of real watersheds but are easier to manipulate and control, may play an important part in model testing and in understanding the significance of parameters. These models, which would be less than an acre in size but much larger and more complex than a soil column or a lysimeter, would allow deliberate departures from homogeneity. The sensitivity of model parameters to such variations could then be established under controlled conditions.

The third aspect of past research, generalization of results for more extensive application, needs much more emphasis. The SCS curve number procedure for estimating direct runoff and the Universal Soil Loss Equation are examples. These techniques were developed before the advent of, or during the infancy of high-speed computers and the models used were accordingly simple. This constraint has been relaxed considerably so it appears that significant improvements could be made. For example, the direct runoff estimation procedure could be improved by incorporating a simple soil moisture accounting instead of the antecedent rainfall index. The functional form of the equation could be changed to more closely approximate results predicted by modern infiltration theory. Results from complex hydrologic models should be used along with experimental results to develop a new procedure for estimating direct runoff.

It should be emphasized that no single model will meet all needs. We need a set of models, involving increasing abstraction, and an objective procedure for selecting the appropriate one for the job at hand.

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CHAPTER 3

CROPLAND EROSION AND SEDIMENTATION

W. H. Wischmeier

Erosion is the wearing away of the land surface by water, wind, ice, or other geological agents. Sediment is defined as solid material, both mineral and organic, that has been moved from its original source by these agents and is being transported or has come to rest on the earth's surface (*66*). Sediment impairs the quality of the water resources in which it is entrained and often degrades the location where it is deposited. It may carry pesticides, toxic metals, and plant nutrients absorbed on the soil particles (*25, 69*).

This chapter documents technical background and methodology for estimating and controlling cropland sediment production. It supplements the material given in Volume I, Sections 3.3 and 4.1. Only sediment from cropland erosion by water was considered pertinent to the purposes of this manual, but literature on wind erosion control is cited (*12, 60, 61, 99, 100*). Observed quantities of sediment from geological erosion and from nonagricultural sources are cited to help portray cropland sediment in its proper perspective, and land classifications pertinent to large-area appraisals of cropland sediment potential are reviewed. Brief overviews of (a) existing erosion research data, (b) the mechanics of the soil-erosion process, and (c) progressive improvements in prediction equations, provide pertinent background information for erosion-control technology. The Universal Soil Loss Equation, soil loss tolerances, and sediment delivery ratios are reviewed as potential tools for pollution-control planning. The major emphasis is on discussions of erosion factors and important features of erosion-control practices.

SEDIMENT SOURCES AND QUANTITIES

Sediment concentrations in rivers of the United States range from 200 to 50,000 ppm, with an occasional concentration as high as 600,000 ppm (*21*). The amount of sediment moved by flowing water has been reported to average at least 4 billion tons a year, with about one billion tons reaching major streams (*19*). Estimates ascribe about 30% of this country's total sediment to geological erosion and about half of it to erosion of agricultural lands (*77*).

Geological Erosion

The erosion that occurs under natural environmental conditions of climate and vegetation, undisturbed by man, is called geological, natural, or normal erosion (*66*). Estimates of annual rates of geologic deposition in the United States range from less than 0.30 to 0.74 ton per acre (*38, 65*). Even at such relatively low rates, a large drainage area will produce large quantities of sediment. The Missouri River's name attests to the turbidity of its waters before it was discovered by Europeans. The rate of erosion under natural vegetation reaches a maximum where the mean annual rainfall is between 10 and 15 inches. Under higher rainfall rates, improved vegetation inhibits erosion; under rates of less than 10 inches sediment-entraining runoff becomes more rare (*29*). Natural erosion over long geologic periods can be quite dramatic, as evidenced by the wearing away of mountains and building up of flood plains.

The more rapid erosion that is primarily a result of activities of man is called accelerated erosion (*66*). Sediment produced by accelerated erosion comes from many sources.

Nonagricultural Sources

Some major nonagricultural sources of sediment are: erosion from construction activities, roadside erosion, stream channel and streambank erosion, scouring of flood-plain land by floodflow, mining and industrial wastes dumped into streams or left in positions susceptible to erosion, and mass wasting from landslides.

In some watersheds, the sediment that originates from these sources may far exceed that from cropland.

A 1969 report by the Secretary of Agriculture and the Office of Science and Technology (*19*) gave the following statistics: During road construction in Scott Run Watershed, Fairfax County, Virginia, sediment at the rate of about 140 tons per acre was produced at the source, and about half of this amount was measured at a downstream gaging station. Erosion losses at rates of 42 to 289 tons per acre per year were measured on bare roadside cuts near Carterville, Georgia, and comparable rates were measured on 35 road cuts in the Baltimore area. As much as 2,000 cubic yards of sediment per square mile of access road has been measured in mountainous country. Sediment from construction activities in urbanizing areas near Lake Barcroft, Virginia, was reported equivalent to 39 tons per acre annually. Studies in southeastern Kentucky showed that sediment yields from strip-mined coal land can be 1,000 times that from forested land; there are about 2.3 million acres of strip-mined lands in the United States. Erosion is a serious problem on at least 300,000 miles of streambank.

Cropland Sediment

Cropland does not produce the greatest amount of sediment per unit of area, but because of the large area involved, our 437 million acres of cropland as a whole produce more sediment than any other source. Annual soil loss from cropland ranges from about one ton to more than 100 tons per acre, depending on the crop system, management practices, rainfall, soil characteristics, and topographic features. A 1967 Conservation Needs Inventory by the USDA (*15, 71*) showed that about half of our country's cropland averages between 3 and 8 tons of soil loss per acre per year, 30% averages less than 3 tons, and 20% averages more than 8 tons. Individual states have published the adjusted inventory data, and the reports are available from state offices of the Soil Conservation Service, USDA (*15*).

Large-Area Estimates of Cropland Sediment Hazards

In 1940, Baver (*7*) listed the major erosion factors as climate, topography, vegetation, soils, and the human factor. The principal influence of climate is the type, amount, and temporal distribution of the rainfall. The human factor includes such items as crop sequence, soil and crop management, and conservation practices. Each of these factors often varies widely within a single watershed or land resource area. All except climate often vary appreciably even among different fields on a single farm. Therefore, soil-loss estimation and control planning are most effective on a local basis, by procedures given in Volume I.

On a large-area basis, the cropland contribution to sediment in streamflow is influenced by: the amount of sediment produced on the cropland (gross erosion), the density of cropland in the drainage area, and the portion of the eroded soil that actually reaches a continuous stream system (sediment delivery ratio).

Gross Erosion

The erosion potential on a relatively homogeneous drainage area can be estimated by using representative soil, cover, and topographic features to evaluate the factors in the Universal Soil Loss Equation. Published maps and standard land classifications also provide helpful information for appraisals of cropland sediment hazard on a large-area basis.

The map given in Volume I as Figure 9 shows relative potential contributions of cropland in the conterminous United States by major land resource areas.

Soil survey maps are the best sources of information on soil characteristics and associated land features. These maps generally include classifications of erosion and land slope. The mapped erosion class is primarily an indication of the extent of prior erosion; quantitative erosion rates are not mapped because of their local nature. The slope class indicates whether the land is nearly level, gently sloping, moderately sloping, strongly sloping, steep, or very steep, but it does not provide information on the slope shapes and lengths.

Land resource units are geographic areas of land, usually several thousand acres in extent, that are characterized by particular patterns of soil (including slope and erosion), climate, water resources, land use, and type of farming (*73*).

Major land resource areas consist of geographically associated land resource units. The 156 major land resource areas of the 48 conterminous states were selected as the basis for mapping hydrologic and erosion-potential data in Volume I. Major characteristics of the 156 individual areas are given in Agriculture Handbook No. 296 (*73*).

Capability classes (*27*) are interpretive soil groupings made primarily for agricultural purposes. The classification begins with the individual soil mapping unit. A capability unit is a grouping of soils that are suited to the same kinds of cultivated crops and pasture plants and have about the same responses to systems of management. A capability subclass is a grouping of capability units having similar kinds of limitations or hazards. Four kinds of limitations or hazards are recognized: erosion, wetness, root-zone limitation, and climate. In the broadest category of capability classification all the soils are grouped in eight classes.

The eight capability classes were briefly described in Chapter 1. A more detailed description is given in Appendix II of USDA Statistical Bulletin No. 461 (*71*). Generally, erosion hazards increase as the capability-class number increases (except Class V), but it is important to recognize that for some areas the higher classifications are due to wetness, root-zone limitations, or climatic limitations rather than erosion. The class and subclass designations, together, provide information about both the degree and the kind of limitation.

Cropland Density

Acreage data for land in each subclass of each of the eight capability classes, by states and several land-use classifications can be obtained from the Conservation Needs Inventory (*15*). These data were used in the development of Figures 6 through 9, Volume I. Wind-erosion limitations were included in the capability subclass data used for Figure 7. The other tables and charts in the erosion sections of Volume I are for water erosion only.

Sediment Delivery Ratio

This is the factor that adjusts the gross sediment estimate to compensate for deposition along the path traveled by the runoff as it moves from a field slope to a continuous stream system. The delivery ratio will be discussed in more detail at the last of this chapter.

CROPLAND EROSION

Erosion Research Data

Measurements of runoff and soil loss from field plots in the United States began about 1917, in Missouri (*64*). Between 1929 and 1933 the U. S. Department of Agriculture established ten Federal-State erosion research stations, in regions where the problem had become most critical. In the next 25 years, erosion plot studies were established at 32 more locations. Precise measurements of precipitation, runoff, soil loss, and related field conditions at the 42 stations in 23 states were continuous for periods of 5 to 30 years (*85*). In 1960, studies were underway on 18 soils. Fundamental studies of erosion mechanics were conducted concurrently and have received increased emphasis since about 1960.

In 1954, the Agricultural Research Service established a national runoff and soil-loss data center at Purdue University. The basic data from more than 10,000 plot-years of erosion studies at 42 research stations were assembled, standarized in units, and transferred to punched cards for summarization and overall statistical analyses (*79*). Data from continuing studies were added annually for analysis with the previously assembled data.

The plot studies and fundamental investigations identified the major erosion factors and provided a wealth of information on erosion mechanics and control. Inherent limitations of the plot data will be pointed out in the discussion of soil loss equations.

Field-plot rainfall simulators are now used to expedite filling voids in existing plot data and field testing of new erosion control concepts and practices. This equipment can simulate the drop sizes and terminal velocities of natural rain at common intensities, apply simulated rainfall on several 75-foot plots simultaneously, and apply identical storms to plots on physically separated soils and topographies (*43*).

The Erosion Process

Soil erosion is a process of detachment and transportation of soil materials by erosive agents (*16*). It is a mechanical process that requires energy. Much of this energy is supplied by falling raindrops. The dead weight of the water falling in 30 minutes of a Midwest thunderstorm may exceed 100 tons per acre. The billions of drops which comprise this 100-ton volume of water strike the soil, if unprotected, at an average velocity of nearly 20 miles an hour. The impact energy during the 30 minutes may exceed 1,000 foot-tons per acre (*93*).

When raindrops strike bare soil at a high velocity, they shatter soil granules and clods and detach particles from the soil mass. Splash action and shallow overland flow transport some of the detached particles directly down the slope and others to implement marks and other small channels, where the more concentrated runoff provides transportation for them. This soil movement is called *sheet erosion* (*6*), or *interrill erosion* (*40*). This type of erosion occurs rather uniformly over the slope and may go unnoticed until much of the productive topsoil has been removed. In sheet erosion, nearly all of the soil-particle detachment is by raindrop impact (*32, 36, 101*).

The erosive potential of flowing water depends on its velocity, depth, turbulence, and type and amount of material it transports (*17*). Water moving down the slope follows the path of least resistance and concentrates in tillage marks, eroded flow channels, and depressions in the natural land surface, where it gains in depth and velocity. Erosion in these flow concentrations is directly related to the hydraulics of the concentrated flow (*40*). The concentrated runoff may remove enough soil to form small but well defined channels, or rills. Rills are often the first readily apparent evidence of erosion, but tillage usually obliterates them.

Rill erosion has been defined as an erosion process in which numerous channels only several inches deep are formed (*66*), and as the erosion occurring in flow channels (*40*). In rill erosion, soil particles are detached by the shearing action of water flowing over the soil surface and by slumping of undercut sidewalls and small headcuts. The detached particles are transported by a combination of rolling, saltation, and suspension. Particles transported by suspension may travel long distances before being deposited on the land surface. The capability of runoff to detach soil material is proportional to the sheer stress raised to a power of approximately two (*17*). Consequently, rill erosion increases rapidly as steeper or longer slopes increase runoff flow depth. Under continued rainfall, sheet erosion continues between the rills. Field soil losses are usually a combination of sheet and rill erosion, and their relative contributions to total soil loss differ with soils and surface conditions.

When water accumulates in narrow channels and, over short periods, removes the soil from this narrow area to depths of 1 to 2 feet, or more, the process is called *gully erosion* (*66*). Gully erosion produces large amounts of sediment but can usually be prevented on cropland.

A soil's inherent ability to resist erosion by rainfall and runoff depends on its physical and chemical properties. Erosion control is accomplished by reducing the mechanical forces of the water acting on the soil particles or by increasing the soil's resistivity to erosion, or both.

Soil Loss Equations

The literature of the past 40 years includes many reports of local erosion studies. These reports may appear to a casual reader as inconsistent, and sometimes incompatible, because of wide differences in the reported results. However, most of these differences can be accounted for by the fact that the rainfall pattern, soil properties, topographic features, and numerous management details occurred at different levels and in different combinations in the various studies.

Plot data predict specific-field soil losses only if the influence of each of the major contributing parameters can be isolated and evaluated relative to the level at which the parameter was present in the study, so that the various influences can be combined in different proportions to simulate other situations. However, effects of rainfall characteristics and soil properties cannot be isolated in a one-location study, where rainfall and soil are either constant for the plot series or vary in unison. Also, many relevant secondary variables cannot be controlled in plot studies. Some of these vary randomly over time. Some differ with seasons, and others, such as rainfall distribution and storm characteristics, show long-term trends at a given location but fluctuate unpredictably for short time periods. The uncontrolled variables interact with controlled variables, and these interactions can substantially bias brief-period research results. Assembling all the available erosion research data at one location for overall statistical analyses (*79*) counteracted many of these limitations. It enabled combining basic data from various locations in analysis designs capable of providing information on the major factor effects individually and on some of the most important interaction effects. It also helped minimize bias of results by random variables.

Mathematical relationships were derived whose basic and theoretical validity has been substantiated by subsequent fundamental research. When these factor relationships are combined in a general soil loss equation, planners can determine what the average annual soil loss rate and the potential soil loss reductions from various alternative crop and management systems are likely to be at specific locations other than that of a plot study.

The most accurate soil loss equation that is now field-operational is the Universal Soil Loss Equation. This equation has been used as an erosion-control planning tool for more than a decade in the 37 states east of the Rocky Mountains and is now used to a more limited extent also in the Western States, Hawaii, and several foreign countries. However, the following brief overviews of four soil loss equations are pertinent to the subsequent discussion of erosion factors.

The Slope-Practices Equation

This initial soil loss equation was developed gradually in the early 1940's. Zingg (*102*) developed factors for the effects of length and steepness of slope. Smith (*62*) added crop and conservation practice factors and the concept of a limiting annual soil loss. Browning and coworkers (*10*) proposed soil-erodibility and management factors for Iowa, but their work was not published until 1947. With the cooperation of program leaders in the North Central Region of the Soil Conservation Service, these initial developments were combined in the Slope-Practice Equation for use throughout the Corn Belt.

This equation used several dimensionless factors to adjust an initial basic soil loss to specific field conditions. Its basic soil loss was the average annual loss from corn-oats-meadow rotations on research plots in the North Central States. Factors for other crop systems were estimated relative to this rotation. The equation had no rainfall factor, and its soil factor was expressed relative to 1.0 for Marshall silty clay loam. Zingg's slope length and steepness exponents (0.6 and 1.4) were used to adjust the soil-loss computations to field slope dimensions.

The Musgrave Equation

In 1946, a national committee, with G. W. Musgrave as chairman, was assembled in Ohio to reappraise the factors in the Slope-Practice Equation and add a rainfall factor. The modified model became known as the Musgrave Equation (*48*). A graphical solution of the equation was published in 1952 for the Northeastern States (*35*).

The 1.75 power of the 2-year, 30-minute rainfall was adopted as the rainfall factor, and Zingg's slope-length and percent-slope exponents were lowered to 0.35 and 1.35, respectively. Annual cover factors were estimated relative to a value of 1.0 for either continuous fallow or continuous rowcrop. A quantitative soil factor was derived by adjusting annual soil losses for effects of rainfall, slope and cover. Subsequent research did not confirm the adequacy of 2-year, 30-minute rainfall as an index of local differences in rainfall erosivity. The lowered slope-length factor was compatible with some early sets of data but too low for others. Numerous plot studies showed that continuous fallow and continuous rowcrop are not interchangeable and that the cover effect of continuous rowcrops is highly variable.

The Musgrave Equation has been widely used for estimating gross erosion from large heterogeneous watersheds. Its highly generalized factor values are more easily assigned to broad areas than are factors based on more specific descriptions of the erosion-influencing parameters. However, erosion hazards are highly localized. For resource-conservation and pollution-control planning, soil loss equations need to reflect local conditions as accurately as possible.

The Universal Soil Loss Equation (USLE)

The Universal Soil Loss Equation (*80, 94, 95*), developed in 1958, overcame many of the deficiencies of its predecessors. Its form is similar to that of the Musgrave Equation, but the concepts, relationships and procedures underlying the definitions and evaluations of the erosion factors are distinctly different (see section on Erosion Factors). The major improvements (*84*) included:

1. More complete separation of factor effects so that results of a change in the level of one or several factors can be more accurately predicted.
2. An erosion index that provides a good estimate of the erosive potential of rainfall and its associated runoff.
3. A quantitative soil-erodibility factor that is evaluated directly from research data without reference to any common benchmark.
4. An equation and nomograph capable of computing the erodibility factor for numerous soils from soil-survey data.
5. A method of including effects of interactions between cropping and management parameters.
6. A method of incorporating effects of local rainfall pattern and specific crop cultural conditions in the cover and management factor.

The Universal Soil Loss Equation computes average annual soil loss as the product of two quantitative factors (soil-erodibility and rainfall-erosivity) and four qualitative factors (*96*). The equation is:

$$A = R\,K\,L\,S\,C\,P$$

where A is the average soil loss, in tons per acre, for the time period used for factor R (usually average annual).

R is the rainfall and runoff erosivity index.

K, the soil erodibility factor, is the average soil loss in tons per acre per unit of R, for a given soil on a "unit plot" which is defined as 72.6 feet long, with 9% slope, continuously fallowed, and tilled parallel to the land slope.

L, the slope-length factor, is the ratio of soil loss from a given length of slope to that from a 72.6-foot length with all other conditions identical.

S, the slope-steepness factor, is the ratio of soil loss from a given percent-slope to that from a 9% slope with all other conditions identical. (In practice, factors L and S are usually combined in a single topographic factor denoted by LS.)

C, the cover and management factor, is the ratio of the soil loss with specified cover and agronomic practices to that from the fallow condition on which factor K is evaluated.

P, the practice factor, is the ratio of soil loss with supporting practices such as contouring or strip-cropping to that with straight-row farming up and down the slope.

The concepts and relationships underlying the evaluations of these factors are reviewed in the discussion of Erosion Factors.

Basic Erosion Models

Basic mathematical models are being developed that combine fundamental principles, concepts and relationships of erosion mechanics, hydrology, hydraulics, soil science, and meteorology to simulate the erosion and sedimentation processes. Substantial progress has been made in developing static and dynamic models capable of predicting spatial and temporal variations in erosion and sedimentation (*14, 17, 40, 52*). To the extent that these simulation models reflect direct and interacting effects of more of the uncontrolled and secondary variables, they will enhance analyses of erosion systems and control practices. These models have not become field operational because additional research is needed to bridge certain information gaps. However, they have already improved the understanding of erosion processes, helped explain some of the seeming inconsistencies in the field-plot data, and improved the accuracy of some of the factor evaluations for the USLE.

The initial basic models have added several important new concepts. One is the treatment of soil detachment by rainfall, detachment by runoff, and transport by runoff, as individual subprocesses that bear substantially different relationships to the erosion factors and that occur in widely differing combinations (*39, 44*). Either detachment capacity or transport capacity can limit erosion at a given site. Another new concept is the separation of rill erosion from interrill erosion (*17, 40*). This distinction will help clarify unexplained differences in the erodibilities of soils and effectiveness of crop canopies. Some soils allow very substantial sheet erosion without rilling; others are much more susceptible to rilling.

EROSION FACTORS

The climatic, soil, topographic, and management parameters that largely determine erosion rates have wide ranges of possible values, or levels, that can occur in any of an extremely large number of possible combinations. The six major erosion factors discussed in this section estimate the effects of different levels of these parameters on soil erosion by water. In a soil loss equation, each factor must be represented by a *number* that reflects the specific local conditions, and all the numbers must be relative to the same, clearly defined, benchmarks. The benchmark conditions for the Universal Soil Loss Equation are free of geographic bounds and are defined as follows.

The benchmark management condition is continuous fallow that receives primary and secondary tillage each spring and is periodically tilled during the summer to prevent vegetation and serious crusting. The tillage operations are up and down the slope. This condition was selected because: (a) continuous fallow is the only condition under which soil effect could be evaluated independently of cover, management, and residual effects, and (b) it is a more constant condition than would exist with any type of cropping. The fact that this condition rarely exists in practice is immaterial because the soil loss computed by the equation as a whole does reflect existing field conditions.

The slope length of 72.6 feet was selected as a benchmark because most of the erosion research plots since 1930 were of this length. It is sufficient for measurement of runoff effect as well as raindrop-impact effect. Slope steepness of 9% was the most representative for the existing plot data. Straight-row farming up and down the slope represents complete absence of support practices. The "unit plot" on which the quantitative soil factor is measured has these benchmark conditions and factors L, S, C, and P have values of 1.0.

The values of factors R, K, L, and S are essentially firm for a particular location and, together, determine the location's characteristic erosion potential. The farmer or planner has no control over rainfall pattern or steepness of the slope. The effective slope length can be reduced by use of terraces or diversions, but this reduction can be classified as a practice effect. Management systems that gradually improve soil structure and increase its organic-matter content can affect its erodibility, but an appreciable change in the soil factor would require many years. Factors C and P, on the other hand, are highly responsive to executed management decisions. Good management and erosion control practices reduce sediment production primarily through their effects on these two factors. The following discussions of the six major erosion factors include the concepts and relationships underlying their definition and evaluation for the USLE.

Rainfall and Runoff Erosivity (Factor R)

Most cropland erosion by water is directly associated with rain events and is influenced both by the rain intensities and by the amount and rate of runoff. The function of factor R is to quantify these interrelated erosive forces. The parameter used to evaluate R must be predictable on a probability basis from meteorological data. It must be definable for specific storms and for specific time periods other than annual, and its seasonal or annual evaluation must be influenced by all significant rains rather than only by annual maxima.

The Rainfall-Erosion Index, EI

The assembled plot data showed that when all factors other than rainfall are constant, storm soil losses from a cultivated field are directly proportional to an interaction term, which is the product of the rainfall energy and the maximum 30-minute intensity. This product is the EI parameter (*80, 93*). The relation of soil loss to EI is linear; therefore, individual-storm values of EI can be summed to obtain seasonal or annual values of the parameter. Frequency distributions of annual, seasonal, or annual-maximum-storm EI values follow the log-normal type of curve that is typical of many hydrologic data(*80*).

Median raindrop size increases as rain intensity increases, to about 3 in/hr, and terminal velocities of free-falling waterdrops increase with increased drop size (*22, 33*). Since the kinetic energy of a given mass in motion is proportional to velocity squared, rainfall energy is directly related to rain intensity. Analyzing published dropsize and terminal-velocity data, Wischmeier and Smith (*93*) derived the equation $E = 916 + 331 \log_{10} i$, where E is the kinetic energy in foot-tons per acre-inch of rain, and i is intensity in inches per hour. The energy of a rainstorm can be computed from recording-raingage data. The storm is divided into successive increments of essentially uniform intensity, and a rainfall energy-intensity table (*93*) derived from the above formula is used to compute the energy of each increment. Thus, the energy of a rainstorm is a function of all its component intensities and rain amount.

In exploratory analyses of data from bare fallow plots, rainfall energy was the best single predictor of associated runoff, but was not a good predictor of soil loss. For sheet erosion, soil detachment is primarily by raindrop impact on the surface, but the capacity of the associated runoff to detach and transport soil material is directly related to its depth and velocity. These are directly related to the maximum prolonged intensity of the storm. Therefore, the erosive potential of a rainstorm is a function of its kinetic energy, maximum prolonged intensity, and their interaction, all three of which are reflected in the EI parameter.

The published rainfall energy-intensity table (*84, 93*) applied the equation given above to intensities up to 10 in/hr. Two recent studies showed that median drop size does not continue to increase when intensities exceed about 3 in/hr (*11, 26*). Therefore, the energy given in the table for a 3 in/hr intensity should be used for all higher intensities as well. This change does not significantly affect EI computation in the United States because prolonged intensities greater than 3 in/hr are too rare to have much effect on average annual EI values.

For computation of average annual EI values, continuous records of from 20 to 22 years are desirable in order to avoid bias by cyclical variations in rainfall pattern (*49*). Erosion index values were computed for about 2,000 locations fairly uniformly distributed over the 37 states east of the Rocky Mountains. By interpolating between the computed point values, lines of equal value (iso-erodents) were plotted on a map that included county lines as references (*82, 96*). The mapped values represent 22-year rainfall records (1937-1958). At stations where 40-year records were available, the 40-year average annual rain amounts generally coincided very closely with the corresponding averages for the 22 years used in development of the iso-erodent map.

The computed annual EI values are reasonably well correlated with the 2-yr, 6-hr rainfall probabilities published by the Weather Bureau (*74*). The relationship is expressed by $EI = 27.38P^{2.17}$, where P = the 2-yr, 6-hr rainfall (*87*). The EI values given in Figure 10a, Volume I, for the 11 Western States were estimated by this equation. Those for the other 37 states were taken from the original iso-erodent map (*82*).

Factor R in the USLE usually equals the pertinent EI value. For prediction of average annual soil loss, it is the annual–EI value available from Figure 10a; for short specific time periods, it is the actual local EI for that period. However, there are two conditions for which the computed EI must be modified to evaluate factor R.

1. Where snowmelt runoff on moderate to steep slopes is significant, the EI value must be adjusted upward to add the erosive effects of this runoff to the R value. The Palouse Region of the Northwest exemplifies this condition. Numerical evaluation of the erosivity of runoff that is not an immediate consequence of rainfall is an area of needed research. Only tentative estimates of the adjustment factor for the Palouse Region are presently available.

2. Experience has shown that on the Coastal Plains of the Southeast, factor R is less than the EI values computed by the standard procedure. This discrepancy may be due to the combination of hurricane-associated storms and flat slopes. The hurricane storms compute very high EI values, but the gentle slopes are soon largely covered by very slowly moving runoff that shields the soil surface from raindrop impact. In a study on a similar soil and slope in the Maumee Basin of Northeastern Indiana, using a rainfall simulator and inflow at the upper end of the plot, drop impact on the soil surface was needed to obtain significant soil loss from a 35-foot plot (*36*). A maximum of 350 for EI values in the Southeast was recently adopted as a temporary measure until research can provide "effective EI" values for these conditions.

Runoff

Surface runoff is not a separate factor in the Universal Soil Loss Equation or its predecessors because: (a) no satisfactory prediction equation for cropland runoff existed, and (b) the respective roles of rainfall and runoff in the erosion process had not been separated in erosion research. Runoff data alone do not predict soil loss. The sediment content of an acre-inch of runoff can range from a mere trace to many tons. For soil loss prediction, the factors in the USLE would need to be combined with the runoff factor, and the runoff would first need to be predicted as a function of essentially the same parameters. Therefore, it was advantageous to relate the factors directly to soil loss in an equation for widespread field use. The EI parameter combines estimates of runoff amount and rate with the potential of the rainfall to detach soil material by drop impact and splash action.

Researchers have recently made good progress in separating rainfall-induced (interrill) erosion from runoff-induced (rill) erosion (*17, 40*). With this separation, a runoff factor added to the soil loss equation should have substantial potential for improved accuracy. An equation that predicts the two types of erosion as separate components of the total soil loss could largely solve the aforementioned problems with factor R in the Northwest and Southeast. Also, some erosion-control practices greatly reduce soil loss without appreciable effect on runoff. Onstad and Foster (*52*) obtained good results from adding a runoff factor to the USLE when using the equation to route sediment through a watershed. However, more research is needed to make this approach field operational.

Soil Erodibility (Factor K)

The susceptibility of a given land area to erosion is a function of all the factors in the soil loss equation, but some soils erode more readily than others even when rainfall, topography, cover and management are identical. Soil erodibility refers to a soil's inherent susceptibility to erosion by rainfall and runoff. This is a function of complex interactions of soil physical and chemical properties. Numerous researchers have measured differences in the erodibilities of a few soils, and some have related erodibility to specific soil properties (*5, 10, 34, 46, 50, 51, 55, 76, 91*). Water infiltration into soils was reviewed by Parr and Bertrand (*54*).

The relation of soil loss to EI is linear, and the average increase in soil loss for each additional unit of EI differs for different soils (*93*). The average soil loss per unit of EI, measured under the previously defined "unit plot" conditions, is the numerical soil-erodibility factor of the USLE. For 23 benchmark soils for which K was measured in long-term plot studies under natural rain, its value ranged from 0.03 to 0.69 (*50, 96*).

Rainfall simulators were used in the Corn Belt, the Southeastern States, and Hawaii to evaluate other soils and obtain soil loss data for study of the relationships of various soil properties to erodibility (*5, 89, 91*). In a Corn Belt study of about 60 soils selected to include a broad range in soil properties, 24 primary and interaction terms were statistically significant in multiple regression analysis of the data (*91*). This illustrates the complexity of the problem, but for practical purposes many of these terms can be neglected either because of relatively small effect or because they are closely related to particle-size distribution, organic-matter content, soil structure or permeability.

Two recent findings were particularly helpful for simplifying the prediction of inherent soil erodibility: (a) that from the viewpoint of erodibility, very fine sand (0.05 - 0.10 mm) would be more properly classified as silt than as sand (*91*), and (b) that percentages of sand, silt and clay must be considered in relation to each other, because of strong interaction between particle sizes. The most informative particle-size parameter in the Corn Belt study was M = % silt(100 - % clay), where the very fine sand is included in the silt fraction. When this parameter was included with organic-matter content, a soil structure index, and the profile permeability class, prediction of the erodibility factor was well within the accuracy needed for field use (*89*). The equation is:

$$K = (2.1 \times 10^{-6})(12 - Om)M^{1.14} + 0.0325(S - 2) + 0.025(P - 3),$$

where Om = percent organic matter, M = the particle-size parameter presented above, S = structure index, and P = permeability class (*92*). Permeability is a profile parameter; the other three pertain to the upper few inches of soil.

The soil-erodibility nomograph presented in Volume I, Figure 11, provides a quick graphic solution to this equation. However, the relationship changes when the silt fraction exceeds 70%. This change is reflected in the nomograph by the bend in the percent-sand curves, but is not reflected in the above equation. The structure-index and permeability-class codings are defined on the nomograph (*89*).

For a few special conditions, the nomograph solution may be modified to improve K-value accuracy: (1) Fragipans and claypans reduce permeability in wet seasons, but do not greatly reduce it for thunderstorms that occur when soil is relatively dry. Separate erodibilities can be computed for dry and wet seasons by using different permeability ratings in the nomograph formula. (2) The mulching effects of stone, gravel, or shale on the surface are not accounted for in the nomograph equation. If used on such soils, it would be applied to mechanical-analysis data for the soil exclusive of the large fragments, and the indicated K value would then be reduced by treating the large fragments as partial mulch cover. (3) The nomograph lacks sensitivity to differences in erodibilities of desurfaced high-clay subsoils, because other chemical properties become important under those conditions. Recent studies showed free iron and aluminum oxides were important for high-clay subsoils but not for most topsoils (*58*).

Standard texture classes are too broad to be accurate indicators of erodibility. Therefore, the K values listed in Table 2a of Volume I are only first approximations. Nomograph solutions will show a broad range of erodibilities within a texture class.

Topographic Features (Factors L and S)

Soil loss per unit area increases as slopes become longer or steeper. The USLE denotes effects of slope length by L and effects of steepness by S. Both are dimensionless and expressed relative to the "unit plot" dimensions defined for factor K. In practice, the two are combined in a single topographic factor denoted by LS.

Slope length is the distance from the point of origin of overland flow to the point where either the slope decreases enough that deposition begins, or the runoff water enters a well-defined channel (*63*). The effect of slope length on runoff per unit area is generally not of practical significance, although there have been instances of statistically significant direct and inverse relationships (*83*). Neither is soil detachment per unit area by raindrop impact greater on long slopes. The effect of slope length is, therefore, primarily due to greater accumulation and more channelization of runoff on the longer slopes. This increases the capability of the runoff to detach and transport soil material.

Factor L in the USLE is dimensionless. For slopes steeper than 4% it is generally computed by the formula $L = (\Lambda/72.6)^{0.5}$, where Λ = slope length in feet and 72.6 feet is the benchmark length. The exponent of 0.5 is the average of values obtained in 10 independent studies in which the observed values ranged from 0.3 to 0.9 (*97*). Field observations indicate that the exponent is probably about 0.3 for slopes of less than 3%, and 0.4 for 4% slopes. Increasing the exponent to 0.6 when slopes exceed 10%, as suggested in Agriculture Handbook No. 282, is of questionable validity. The higher exponents observed in the length-effect studies were associated with plowed-out bluegrass sod or abnormally severe rain events, on slopes that did not exceed 10%. Both L and S are believed to be influenced by density of cover, soil erodibility, and rainstorm characteristics, but existing data are inadequate for mathematical evaluations of these interaction effects.

There have been field indications that the slope-length exponent becomes smaller for extremely long slopes. This is logical because slopes approaching a thousand feet in length would rarely have a constant slope steepness along their entire length, and upslope depositional areas would be likely.

Slope steepness affects both runoff and soil loss. In the assembled plot data, runoff from small grain tended to increase linearly with increases in slope. For row crops the increase was curvilinear, increasing at an increasing rate (*83*). Soil loss increases more rapidly than runoff as slopes steepen.

The combined effects of length and steepness for uniform slopes were shown in Table 3, Volume I. The table was derived by the formula

$$LS = \left[\frac{\Lambda}{72.6}\right]^m \left[\frac{430 \sin^2 \theta + 30 \sin \theta + 0.43}{6.574}\right]$$

where m = 0.5 if the slope is steeper than 4%, 0.4 for 4% slopes, and 0.3 for slopes of 3% or less; and θ = the angle of slope.

The last quantity in this equation is an unpublished conversion of an earlier formula (*96*) to an expression in terms of the sine of the angle of slope. Within the range of the research data, the two forms are equally accurate, but an expression in terms of sin θ is more logical and computes more realistic values when extrapolated to steeper slopes.

The research data used to derive relationships of slope length and steepness to soil loss were from plots not longer than 270 feet and slopes not steeper than 18 percent. The extrapolated values shown in Table 3 of Volume I for slopes that exceed these dimensions, although speculative, are the best estimates presently available. Soil loss estimates for slopes steeper than about 30% are potentially subject to considerable error. Research on steep slopes is a major need.

Shape of the slope is also important. When a slope steepens or flattens significantly toward the lower end, or is composed of a series of convex and concave segments, its overall average gradient and length do not correctly indicate the topographic effect on soil loss. An irregular slope can be viewed as a series of segments such that the gradient within each segment can, for practical purposes, be considered uniform. The segments cannot be evaluated as independent slopes when runoff flows from one segment to the next. However, the amount of soil detached on each segment can be computed by a recently published formula (*18*) and summed for the entire slope length. For each segment, the effective slope length is the distance from the top of the overall slope to the foot of the particular segment.

If the segments are selected so that they are also of equal length, the slope-effect table for uniform slopes can be used with appropriate adjustment factors for position of the segment on the overall slope. For most field slopes, three segments should be sufficient. The procedure is as follows (*87*):

Ascertain the percent slope for each segment. Enter the slope-effect chart or table with the total slope length and read the LS value corresponding to the steepness of each of the three segments. Multiply the chart LS value for the upper segment by 0.58, the middle-segment value by 1.06, and the lower-segment value by 1.37. The average of the three products is a good estimate of the effective LS value for that slope. The three products also indicate the relative magnitudes of soil loss on the three slope segments. (If two segments are sufficient, use the multiples: 0.71 and 1.29. For four segments: 0.50, 0.91, 1.18, and 1.40. For five segments: 0.45, 0.82, 1.06, 1.25, and 1.42.) (*87*).

Cover and Management (Factor C)

The ability of a soil to resist the erosive forces of rainfall and runoff is profoundly influenced by the direct and residual effects of vegetation, crop sequence, management, and agronomic erosion-control practices. The effects of cropping and management must be estimated in combination, because of many interrelated variables. Nearly any crop can be grown continuously or in any one of numerous rotations. The sequence within a system can be varied. Crop productivity can be low, or it can be high. Crop residues can be removed, left on the surface, incorporated near the surface, or plowed under. The amount of residues can vary from scattered pieces to complete surface cover. The crop can be planted in a pulverized and smoothed seedbed, in a rough and cloddy seedbed, or with extremely little soil disturbance. It can be intertilled after emergence, or the weeds can be controlled with chemicals. The effectiveness of crop-residue management will depend on the amount of available residue. This, in turn, depends on the rainfall distribution, the fertility level, and various management decisions made by the farmer. Also, the residual effect of meadow sod depends on the type and quality of meadow, on how the succeeding seedbed was prepared, and on the length of time elapsed since the sod was turned under. The erosion-reducing effectiveness of a crop system depends on how the levels of all these variables, and others, are combined on the field.

Factor C in the Universal Soil Loss Equation is the ratio of soil loss from land cropped under specified conditions to the corresponding loss from clean-tilled, continuous fallow (*96*) and therefore includes the effects of all these variables. If the actual soil loss equals the potential loss predicted by the product of factors R, K, L, and S, factor C=1. This would be clean-tilled continuous fallow or land where mechanical desurfacing has removed all of the surface vegetation and most of the root zone. Where there is any vegetative cover, where the upper layer of soil contains significant amounts of roots or plant residues, or where cultural practices increase infiltration and reduce velocity, soil loss is less than the product RKLS. Factor C brings this reduction into the soil loss computation. On cropped land, C ranges from about 0.60 downward to less than 0.01. This great flexibility in the value of C is extremely important to erosion-control planners. If C is reduced, soil loss is reduced by the same percentage.

The canopy protection of crops varies widely for different weeks or months in the crop year. The overall erosion-reducing effectiveness of a crop depends on how much of the erosive rain falls while the crop provides the least protection. The correspondence of periods of highly erosive rainfall with periods of good or poor vegetative cover differs appreciably between geographic regions. Therefore the C value for a particular crop system will not be the same in all parts of the country. A field-tested routine is available for computing site C-values that reflect the net effect of the interrelated crop and management variables in whatever combination they occur at the site and in relation to the local rainfall pattern.

The entire rotation cycle is divided into a series of cropstage periods so defined that cover and management may be considered approximately constant within each period. The five cropstage periods are defined as follows, for each crop-year in the system (*81, 96*):

Period F - Rough fallow. Turn plowing to final seedbeed preparation. (No-plow systems omit this period.)

Period 1 - Seedling. Seedbed preparation to 1 month after seeding.

Period 2 - Establishment. The second month after spring or summer seeding. For fall-seeded grain, this period extends to about May 1 in the Northern States, April 15 in the Central States, and April 1 in the Southern States.

Period 3 - Developing and maturing crop. End of period 2 to harvest.

Period 4 - Residue or stubble. Crop harvest to plowing or new seeding. (When meadow is seeded with small grain, period 4 ends about 2 months after grain harvest. The vegetation is then classified as established meadow.)

Probable calendar dates for the events that begin the successive periods are selected on the basis of local climate and farm practice. The fraction of the annual EI that normally occurs in that locality during each cropstage for each of the crops in the rotation is determined from the applicable EI-Distribution Curve in Agriculture Handbook No. 282. These fractions are multiplied by the corresponding soil loss ratios from Table 2 in the same handbook. The sum of the products obtained for the cropstage periods in any one year is the C-value for that particular crop in that system. The crop-system C-value is the sum of all the partial products, divided by the number of years in the system. This procedure has been used by the Soil Conservation Service to develop local C-value tables that are available from their state offices. The illustrative C values given in Table 4 of Volume I were also derived by this procedure, but the seeding and harvest dates and the EI-distribution data were generalized and are not precise for any particular location.

The 33 regional EI-distribution curves in Agriculture Handbook No. 282 were derived from the 22-year rainfall records used to develop the iso-erodent (R-value) map (*82*.) Corresponding data for the 11 Western States and Hawaii are presently available only as tentative estimates.

The soil-loss-ratio table (*96*) was derived from analysis of more than 10,000 plot-years of erosion data. The data in this table are percentages of soil loss from the indicated combinations of cover and management to corresponding losses from continuous fallow. The table has limitations that need to be recognized. The "minimum tillage" classification applies only to plow-based systems in which disking and smoothing are omitted. A partial list of ratios for no-plow systems that retain some or all of the residues on the surface was published in 1973 (*86*). The ratios for corn in cropstage 4, residues left, are for stalk cover as left by the picker. Shredding the stalks provides more complete cover and reduces the soil-loss ratio. Some of the crop systems in the Western States and Hawaii are not represented, but approximate values for these systems are now available from the Soil Conservation Service, Western Technical Center, Portland, Oregon. Approximate C-values for range, woodland, and idle land were published in 1974 (*87*).

Practices that depend on small rates of residue and/or tillage-induced surface roughness for erosion-control effectiveness will be ineffective if slopes are excessively long. The precise length limits for various mulch rates, slope steepnesses, kinds of soil, and rainfall patterns have not been determined. Investigations by the Agricultural Research Service are underway to improve or verify the approximate limits given in Table 13, Volume I.

Supporting Practices (Factor P)

This factor is similar to C except that P accounts for additional effects of practices that are superimposed on the cultural practices, such as contouring, terracing, diversion, and contour stripcropping. Approximate values of P, related only to slope steepness, were listed in Table 5, Volume I. These values are based on rather limited field data, but factor P has a narrower range of possible values than the other five factors. Influences of type of vegetation, residue management, rainstorm characteristics, and soil properties on the value of P have not been evaluated to the point of predictability.

EROSION CONTROL METHODS

Specific types of erosion-control practices were discussed in Section 4.1 of Volume I. These discussions included general information on the advantages, limitations, and variability of each type of practice. The indicated percentages of reduction in soil loss were based on C values estimated from all the available data rather than on results of any specific local experiment. This section discusses principles and relationships that determine the effectiveness of erosion-control practices.

If surface runoff can be eliminated, movement of sediment from a field will be insignificant. Breakdown of soil aggregates by raindrop impact, rearrangement of soil particles, and surface sealing can occur before runoff begins, but very little sediment will leave the field unless surface runoff is available to transport it. Land treatments that result in a deep, fertile topsoil, a high level of organic matter, good tilth, and good vegetative cover increase infiltration and reduce runoff. These conditions may completely eliminate surface runoff from moderate rainstorms on some areas. Generally, however, where the rainfall is adequate for crop production some of it falls at intensities greater than the soil can infiltrate even when well managed, and runoff occurs.

Land treatments that increase infiltration and the capacity of the soil to store water will reduce small-watershed flooding that results from short, intense rains during the growing season. However, when the soil becomes saturated to a considerable depth, as is often the case in major flood periods, cultural practices have much less effect on runoff. Erosion-control practices must also reduce the shear stress and transport capacity of the runoff. This means reducing runoff amounts, velocities, and depths and dissipating the flow energies on plant residues rather than on the soil surface.

Erosion-control practices rely primarily on five means of reducing erosion: 1) vegetation, 2) plant residues, 3) improved tillage methods, 4) residual effects of crops in rotation, particularly systems that include grass and legume meadow, and 5) mechanical supporting practices. A sixth potential approach would be use of chemical soil stabilizers, but they have not yet become economically feasible for field use.

Vegetation

Vegetation (a) intercepts rainfall and thereby reduces runoff and soil-particle detachment by drop impact, (b) increases the soil's water-storage capacity through transpiration, (c) retards erosion by decreasing runoff velocity, (d) physically restrains soil movement, (e) improves aggregation and porosity of the soil, and (f) increases biological activity in the soil (*59*).

Crop Canopies

Leaves and branches that are not in contact with the soil reduce runoff from small rains but have relatively little influence on the amount and velocity of runoff from prolonged rains. In 542 plot-years of conventionally planted corn, the average runoff per thousand foot-tons of computed rainfall energy was only about 15 percent less in cropstage 3 than before canopy had developed (*90*). But *soil loss* per EI unit from a field of clean-tilled 90-bushel corn is about 60 percent less in cropstage 3 than in cropstage 1 (*96*, Soil Loss Ratio Table), primarily as a result of raindrop interception by the canopy. Water drops from canopy may regain appreciable velocity, but usually not the terminal velocities of free-falling raindrops. Therefore, canopy reduces rainfall erosivity by reducing its impact energy at the soil surface. The amount of reduction depends on its height and density. Canopy effect can be viewed as a reduction in the "effective" EI of the rainstorms and as such can be directly computed for specific situations.

Figure 1 shows the ratios of effective EI's computed for several drop fall heights to the EI of unintercepted rainfall (*88*). Percent cover was defined as the percentage of the total ground area that could not be hit by vertically falling raindrops because of the canopy. Soil loss reductions due to canopy over a bare soil should be approximately proportional to the reductions in effective EI. Figure 1 assumes a median dropsize of 2.5 mm for both the rain and the droplets formed on the canopy. Where rainfall is characteristically of low intensity and small drops, canopy effect would be less.

All crops develop some canopy, but this may require several months, and most or all of it may be lost with the crop harvest. Good soil-fertility management and narrow row spacing hasten the development of a protective canopy. Crop sequences can be selected that substantially reduce the length of time between successive plant covers, and early seeded winter cover crops can provide interim cover.

Vegetation At The Soil Surface

Stands of grass or small grain are much more effective than a raised canopy. Much of the rain that such vegetation intercepts moves down the blades and stems to a point so near the ground that the droplets regain no appreciable energy. The dense vegetation at the soil surface also reduces runoff amount and velocity and physically restrains soil movement. For about 5,000 plot-years of data, runoff from small grain averaged 9 percent of total precipitation, in contrast to 12 percent for row crops (*83*). Meadow averaged 7 percent. The soil-loss-ratio table shows that soil loss from established small grain averages less than half of that under a canopy of conventionally planted corn. Soil loss from a good quality grass and legume meadow is generally negligible.

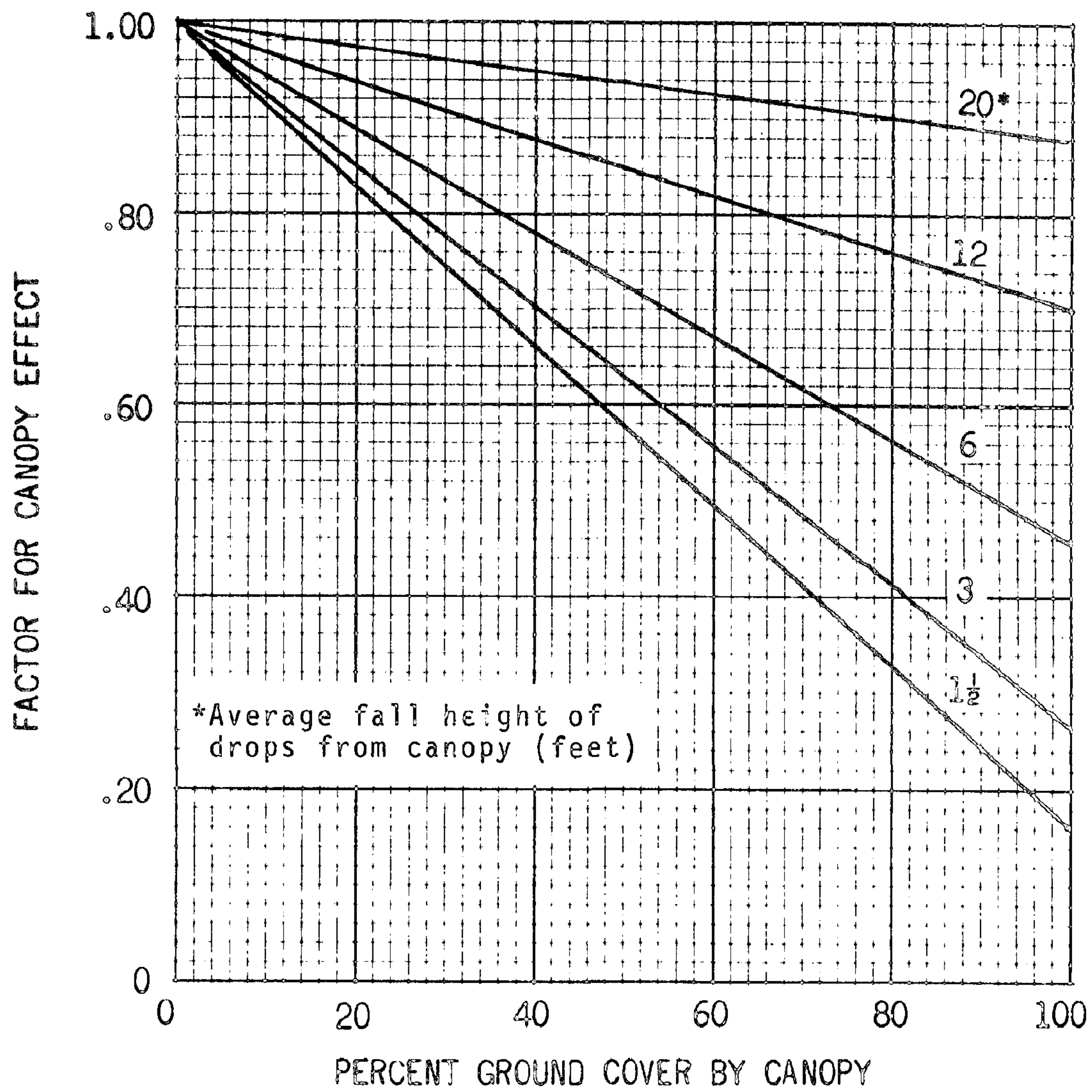

Figure 1.–Effect of crop canopy on effective EI.

Plant Residues

With one-crop systems, crop residues can supply very effective cover during the approximately 8 months from harvest until the next crop develops full cover. Stalky residues such as corn and grain sorghum provide more effective cover when shredded than when left partially standing. Complete residue cover at the soil surface virtually eliminates raindrop impact on the soil, greatly reduces the detachment capability and transport capacity of runoff, and usually increases infiltration. Runoff through or over a complete cover of residue mulch is very low in sediment content.

For rowcrop production, high residue rates are most fully utilized in the no-till systems. The seeds are planted in narrow slots opened through the residue by a fluted coulter or other device, without tillage. No-till planting for corn has been very highly effective in chemically killed meadow or small grain, in grain stubble, in chopped rowcrop residues, and in winter-cover crops. However, the practice is not adaptable to all fields (see Section 4.1, Volume I).

Reductions in soil loss by various rates of straw mulch tested under simulated rain have varied with soil type and surface conditions (*1, 4, 9, 28, 37, 42, 45, 70*). Figure 2 shows the average relation of soil loss with various rates of mulch to corresponding losses with no mulch, as observed on 35-foot cropland slopes subjected to 5 inches of simulated rain in two 1-hour storms (*86*). How much the slope length or steepness could be increased before unanchored mulch would be undercut or transported by the flowing water has, however, not been fully determined. Mulch applied on plowed and disked surfaces of silt loam soils with slopes of 3 to 5 percent substantially reduced runoff, but mulch on a 15% slope of untilled loam from which oat stubble had been removed with a scraper had no significant effect on amount of runoff. Under both conditions, however, one ton of straw per acre reduced the velocity of the runoff by about 60 percent (*45*).

Partial incorporation of the residues by shallow tillage, such as disking, reduces the percentage of surface cover and loosens some of the soil for easy detachment. The residues are then less effective than equal quantities left undisturbed on the surface. In a test under 5 inches of simulated rain, no-till planting without prior disking reduced soil loss 83 percent in contrast to a 73 percent reduction with similar planting after disking (*86*). The amount that soil-loss is increased by shallow tillage will vary in relation to initial amount of residue and how much is covered by the tillage.

Moldboard plowing inverts the upper 6 to 8 inches of soil and usually covers virtually all of the residue. The surface is then quite susceptible to erosion. However, even with annual turnplowing, leaving the residues on the field is far better than removing them. Regular incorporation of crop residues by plowing gradually increases the amount of organic materials in the soil and improves water intake and soil structure. For 82 plot-years of continuous corn with residues removed each fall, runoff during the seedling and establishment months averaged 83% of corresponding losses from fallow; for 50 plot-years in which the residues were plowed down each year, runoff in those months aver-

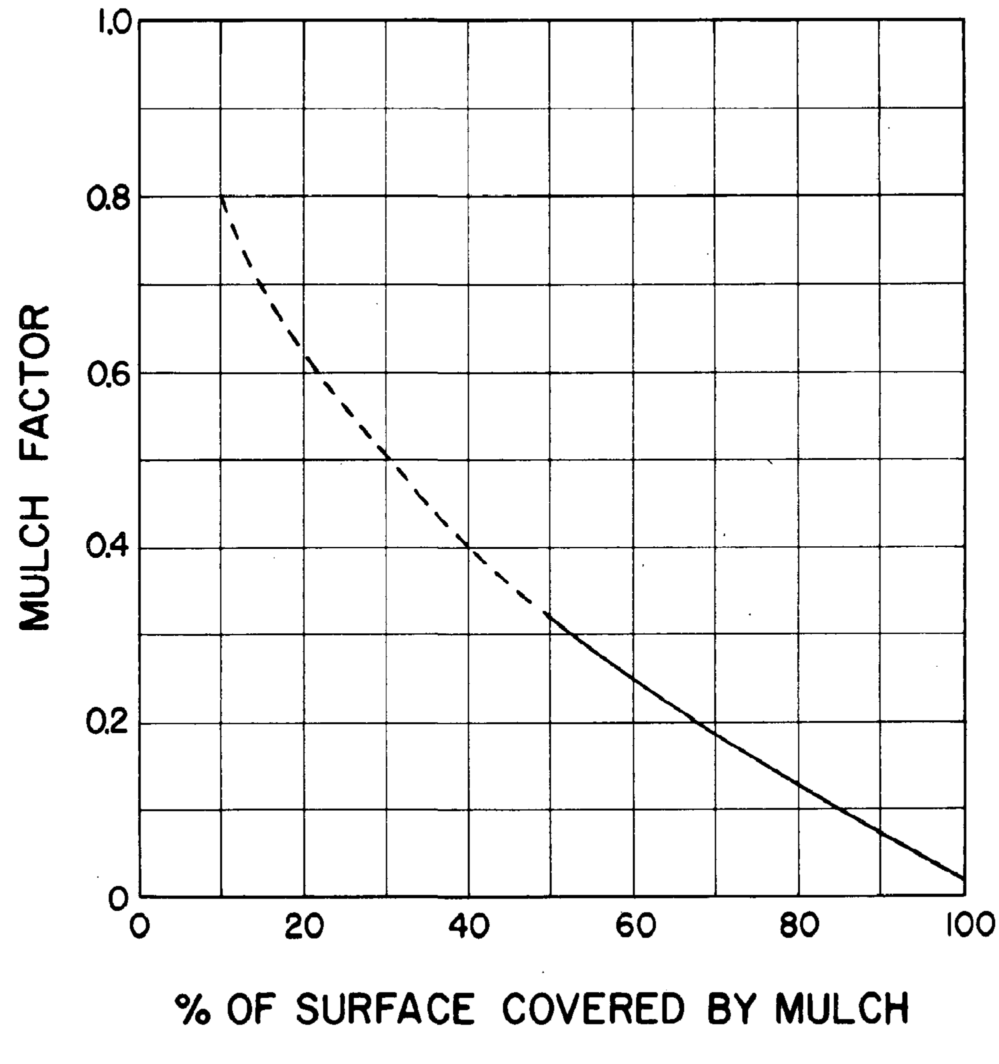

Figure 2.–Effect of plant-residue mulch on soil loss.

aged 51% of that from fallow (*90*). Annual soil loss was about 20% less where residues were incorporated than where they were removed.

Improved Tillage Methods

That influence of a tillage practice on the disposition of crop residues is extremely important for erosion and sediment control is evident from the preceding discussion. The surface microtopography and condition of the soil after tillage also strongly influence the amount of soil erosion. Roughness of the soil surface and porosity of the tilled layer are important parameters in describing the structure of a tilled soil (*31*). Rough surfaces detain considerable quantities of water in microdepressions until they can enter the soil, and the porous soil layer offers a channel system to funnel water throughout the tilled layer. Random roughness reduces runoff velocity, and the water that is temporarily ponded in the depressional areas shields portions of the soil surface from particle detachment by raindrop impact. Also, some of the sediment detached by raindrop impact on the exposed surfaces is deposited on the ponded surfaces. Porosity and roughness are influenced by the type of tillage and by the water content of the soil at the time of tillage. Pulverizing the soil increases erosion by increasing the soil's detachability and increasing the amount and rates of runoff.

When soil dries after a rain that has broken down its surface structure and washed fines into soil voids, crusts develop that are strong enough to reduce seedling emergence (*20, 47*). Surface seals and crusts reduce water intake by the soil and substantially increase erosion (*41, 78*). Rough soil surfaces tend to concentrate the dispersed material in the microdepressions and leave the peaks more porous, but mulches are more effective for preventing surface seal.

Soil compaction by heavy equipment can hamper root and plant development and thereby increase soil erosion. Conservation tillage practices generally require less use of heavy equipment on the field. Aspects of soil compaction were recently summarized by the American Society of Agricultural Engineers (*2*).

Larson (*30*) points out that the secondary soil aggregates around the seed and seedling roots must be small enough to prevent undue drying of the soil, must provide sufficient soil-seed or soil-root contact for moisture transfer, must provide adequate aeration, and must not be so finely divided as to encourage surface crusting or mechanical impedance when dry. However, the area between the rows, which he designates as the water management zone, may be rough and cloddy or may be left untilled under a residue mulch.

Conventional tillage includes primary and secondary tillage operations normally performed in preparing a seedbed for a given crop grown in a given geographic area. Where the term is used in this manual as a basis for comparisons, it includes moldboard plowing and several disking and smoothing operations.

Minimum tillage is the minimum soil manipulation necessary for crop production under existing soil and climatic conditions. The term is often loosely applied to any system with fewer operations than a conventional system, but it is most accurate when applied to plow-plant and wheeltrack-plant systems, in which the field is plowed but secondary tillage is omitted. These systems are most effective for erosion control when the rowcrop follows one or more years of meadow, and before the clods disintegrate.

Conservation tillage includes tillage systems that create as good an environment as possible for the growing crop and that optimize conservation of the soil and water resources, consistent with sound economic practices. Conservation tillage includes maximum or optimum retention of residues on the soil surface and use of herbicides to control weeds (*98*)

No-till is a system whereby a crop is planted directly into a seedbed untilled since harvest of the previous crop.

Residual Effects of Previous Crops

The benefits of crop rotations for minimizing periods of little or no vegetative cover were pointed out earlier, but crops and management practices also have residual effects that influence soil erodibility under succeeding crops.

Sod-based Rotations

The greatest residual effects are derived from grass and legume meadows. In data assembled from conventional seeding and tillage practices, soil losses from corn following meadow ranged from 14 to 68 percent of corresponding losses from corn on adjacent plots in meadowless systems. Grass and legume mixtures were more effective than legumes alone. The erosion-control effectiveness of rotation meadows turnplowed before corn planting was, in general, directly proportional to the quality of the meadow, as measured by hay yields. Erosion reduction was greatest during the fallow and corn-seeding periods and decreased gradually for about 2 years. The effects of well-managed long-term meadows were still apparent in the third year. When second-year hay yield exceeded the first, 2-year meadows were more effective than one-year meadows, but when allowed to deteriorate in the second year they were less effective (*81*).

Meadowless Systems

Crops that are not sod forming also have beneficial residual effects on soil erodibility, but they are much less pronounced than those of grass and legume meadows. Corn generally leaves the soil less erodible than soybeans but more erodible than good quality small grain. All crop systems have beneficial residual effects relative to continuous fallow; brief periods of fallow in a rotation are not as erodible as continuous fallow. Removal of the crop residues year after year gradually reduces soil organic matter and adversely affects soil tilth. One-time incorporation of residues by moldboard plowing had little effect on infiltration or erosion, but repeated incorporation year after year had very substantial effects (*90*).

Mechanical Support Practices

These are mechanical erosion-control practices used when slopes are too long or too steep for agronomic practices alone to control erosion.

Contouring

Furrows made by plowing, planting, and cultivating form natural channels in which runoff accumulates. If the tillage is up and down slope, the shear stress of the runoff increases as the slope of the furrows increases, and erosion may be serious.

In contouring, tillage operations are carried out as nearly as practical on the contour. The general rule is to lay out guidelines which assure that all tillage is within a gradient limit of 1 to 2 percent (*59*). On gently sloping land, contouring will reduce the velocity of overland flow by channeling it around the slope. Contoured ridge or lister planting substantially increases the storage capacity of the furrows and permits storage of large volumes of water. When contouring is used alone on steep slopes or under high rainfall intensities and soil erodibility, the hazard of gullying is increased because row breakovers may release the stored water. Breakovers cause cumulative damage as the volume of water increases with each succeeding row. If the contour lines are not carefully laid out and rows are allowed to cross natural depressions at gradients much greater than 2 percent, adverse results of breakovers may completely offset the beneficial effects of contouring. The effectiveness of contouring is also impaired by decreased infiltration capacity due to surface sealing, and by reduction in depression storage after tillage operations cease and the soil settles (*59*).

Graded Rows

Graded rows are land-formed to a precise gradient. This improves surface drainage and decreases the likelihood of row breakovers.

Contour Stripcropping

Alternating contoured strips of sod with strips of row crops is more effective than contouring alone. The sod strips serve as filters when rows break, and much of the soil washed from a cultivated strip is filtered out of the runoff as it spreads within the first several feet of the sod strip (*64*). In the Mormon Coulee near LaCrosse, Wisconsin, some fields are reported to have been cropped in strips for more than 70 years. Where the strips were on the contour, or nearly so, erosion control was good. Where the strips were sufficiently off-contour to give row slopes of 5 percent or more, soil losses from flow of runoff down the rows were high (*8*).

Systems with alternate contoured strips in meadow reduce soil loss to about half of that from the same rotation with contouring alone. Three-year rotations of sod, small grain, and row crop were slightly less effective. Alternate strips of fall-seeded grain and row crop have effected some reduction in soil loss, but alternate strips of spring-seeded grain and corn on moderate to steep slopes have not proved more effective than contouring alone (*64*).

Buffer stripcropping is a practice in which strips of grass are laid out between contour strips of crops in the regular rotations. The grass strips may be irregular in width and may be placed on critical slope areas in the field (*59*).

Terracing

Terracing with contour farming is more effective for erosion control than stripcropping, because it divides the slope into segments with lengths equal to the terrace spacing. With stripcropping or contouring, the entire field slope length is the effective length. With stripcropping, the saved soil is largely that deposited in the sod strips; with terracing the deposition is in the terrace channels and may be as much as 80 percent (*96*) of the soil moved to the channel. Erosion control between terraces depends on the crop system and other management practices; with stripcropping, an effective sod-based rotation is built into the system.

If a control level is desired that will maintain soil movement between the terraces within the soil-loss tolerance limit, the P factor for terracing should equal the P factor for contouring. However, if the soil loss equation is used to compute gross erosion for watershed-sediment estimation, a terracing P factor equal to 20 percent of the contour factor is warranted *(96)*. Since terracing shortens the slope length, it also reduces the between-terrace soil loss by decreasing the topographic factor. Dividing a slope that is steeper than 4% into n equal segments divides the value of factor L by $\sqrt{n}$. In the table of P values given in Volume I, this reduction was included in the P_t factor for convenience.

The two major types of terraces are the bench terrace and the broadbase terrace *(59)*. Broadbase terraces are broad-surface channels or embankments constructed across the slope of rolling land. They may be either channel type or ridge type. The primary purpose of the graded or channel-type terrace is to remove excess water in such a way as to minimize erosion. The primary purpose of the level or ridge-type terrace is moisture conservation; erosion control is a secondary objective. The channel is level and is sometimes closed at both ends to assure maximum water retention. The Zingg conservation bench terrace is designed for use in semiarid regions for moisture conservation. It consists of an earthen embankment and a very broad flat channel that resembles a land bench.

The steep-backslope terrace is constructed with a backslope of 50% or steeper, which is kept in grass *(8)*. It may be either a graded or a level terrace. Parallel grass-backslope terraces with subsurface drains are now gaining popularity. They release the excessive water slowly and are also better adapted to use of large farm implements than graded or level terraces.

SEDIMENT DELIVERY RATIOS

The sediment delivery ratio is the parameter that bridges the gap between upslope erosion data and drainage-area sediment yield. The sum of the soil-loss estimates for the individual tracts constituting a drainage area approximates the quantity of soil moved from its original general position. To compute drainage-area sediment yield, this estimate must be adjusted downward to compensate for deposition in terrace channels, in sod waterways, in field boundaries, at the toe of field slopes, in depressional areas, and along the path traveled by the runoff as it moves from the field to a continuous stream system or lake *(96)*. Sediment additions from sources along this path must also be taken into account. Further changes in sediment content of runoff water will occur during the stream transport phase. The Universal Soil Loss Equation computes gross sheet and rill erosion, but it does not compute deposition. Nor does it compute sediment from gully, streambank, and channel erosion. The sediment delivery ratio provides a method of accounting for the sediment losses and gains that occur below the areas where the USLE is applied.

The delivery ratio is usually estimated from natural drainage-area parameters and therefore does not account for deposition in terrace channels or in constructed settling basins or traps. The amount of sediment deposited in these man-made devices near the sediment source is subtracted from the computed field erosion to obtain the gross-erosion estimate to which the delivery ratio is applied. Two methods of defining the delivery ratio will be discussed.

Delivery Ratios for Dealing with Downstream Sediment Problems

For this purpose, the delivery ratio is defined as the ratio of sediment delivered at a given point *in the stream system* to the gross erosion from all sources in the watershed above that point. Guides for estimating this ratio were given in Volume I, section 3.3c. The source of most of the information presented there was the Sedimentation Section of the National Engineering Handbook developed by the Soil Conservation Service *(72)*. The approximate delivery ratios that were listed relative to watershed size were obtained from the relationship curve derived from published and unpublished data assembled by L. C. Gottschalk, G. M. Brune, J. W. Roehl, R. Woodburn, S. B. Maner, L. H. Barnes, and L. M. Glymph and presented in the Engineering Handbook. This curve relates the delivery ratio to the negative 0.2 power of drainage-area size. There have been indications that the 0.1 power would be more accurate for large drainage areas *(3)*.

Analyzing data from 14 Texas Blackland Prairie drainage areas that ranged from 0.42 to 97.4 square miles, Renfro *(57)* computed delivery ratios ranging from 0.62 for a drainage area of 0.5 square mile to 0.28 for an area of 100 square miles. These are significantly larger than would have been estimated from the SCS general relationship curve, and emphasize the need to consider the other factors listed in Volume I as well as watershed size. Several other relevant publications are listed in the literature citations (*3, 13, 23, 24, 53, 56, 67*).

Delivery ratios derived on this basis are more appropriate for dealing with downstream sedimentation problems than for estimating the amount and composition of cropland sediment that reaches a continuous stream system. However, they are presently more directly available than those discussed below and can be helpful also for the latter purpose.

Delivery Ratios for Purposes of this Manual

For evaluation of cropland contributions to sediment in stream systems, the delivery ratio should be defined as the ratio of sediment delivered *at the place where the runoff enters a continuous stream system or lake* to the gross erosion in the drainage area above that point. It will then not be biased by sediment-content changes that occur during the stream transport phase. Where this ratio is known or can be closely approximated from drainage-area parameters, multiplying it by the computed gross erosion will estimate the amount of sediment delivered to the stream system.

No general equation for sediment delivery ratios as a function of drainage-area parameters is presently available. A generally applicable upslope-deposition equation is a major research need. However, guides for approximating the average delivery ratio for a particular drainage area are available. The ratio can approach a value of 1.0 for a particular field if the runoff drains directly into a lake or stream system, with no obstructions and no flattening of the land slope. On the other hand, a wide expanse of forest duff or dense vegetation below the eroding area may filter out essentially all of the sediment except some of the colloidal material. These are the extremes.

Anything that reduces runoff velocity (reduction in slope steepness, physical obstructions such as ridges or living or dormant vegetation, ponded water, etc.) reduces its capacity to transport sediment. When the sediment load exceeds the transport capacity of the runoff, deposition occurs. The observed sediment reductions by terracing or contour stripcropping are examples of the potential magnitude of upslope deposition. More than 80% of the soil eroded between terraces may be deposited in the terrace channels because of the large reduction in runoff velocity due to the terraces. Most of the soil eroded from cultivated strips has been observed to be deposited in the sod strips when contoured strips of sod were alternated with equal-width strips in cultivated crops.

Relative to the sediment-source area, the delivery ratio will generally be directly related to amount of runoff and inversely related to soil particle size. Relative to the land between the source area and the stream system, the ratio will be directly related to slope steepness and amount of channel-type erosion, and inversely related to: distance of the source area from the stream system or lake; density of vegetation at ground level; and number of flow obstructions such as field boundaries, culverts, etc.

The delivery ratio for a given drainage area will not be constant for all runoff events, because the depth and velocity of runoff will differ with storm size and antecedent surface conditions. These differences will not only affect transport efficiency; a major runoff event may also pick up some of the sediment deposited enroute to the stream or lake in prior events. The average delivery ratio for a drainage area can be estimated more closely and should suffice for estimates of long-term average sediment yields.

TOLERANCE LIMITS

This section discusses merits and limitations of several alternative methods of defining soil loss or sediment limits, as background information for those who may be involved in developing state sediment control standards. Soil loss limits used to illustrate points are not intended as specific recommendations.

Optimum soil-loss limits for preservation of cropland productivity may differ substantially from optimum sediment standards for control of runoff pollution from nonpoint sources. The underlying considerations are quite different, and specific differences must be recognized. Standards will be most beneficial when they achieve both objectives with the least possible adverse effect on production of food and fiber.

Tolerances for Preservation of Cropland Productivity

Tolerance limits on average annual soil loss have been used in this country for a quarter century to guide soil conservation planning. Limits ranging from 2 to 5 tons per acre are applied to individual field slopes. Experience has shown these limits to be feasible and generally adequate for preservation of high productivity levels. The 2 to 5 ton tolerances represent the collective judgment of soil scientists in the Soil Conservation Service, Agricultural Research Service, and State agricultural experiment stations in the 1950's. Factors considered in defining these limits were published by the Soil

Conservation Service in reports of five regional soil loss prediction workshops held from 1960 to 1962.

One of the major considerations was longtime maintenance of adequate soil depth for good plant growth. The rate of natural soil renewal for mature soils has been hypothesized to balance the rate of erosion under natural conditions, without influences of man (*65*). Since erosion in excess of renewal rates reduces soil depth, shallow soils were assigned lower tolerance limits than those for deep soils with subsoil characteristics favorable for plant growth. Prior erosion was a factor because of its effect on the soil profile. Other considerations included the prevention of field gullying, sedimentation problems, seeding losses, soil organic matter reduction, and plant nutrient losses. Research directed to precise definition of soil loss tolerances (*65, 68*) has been extremely limited.

Tolerances for Sediment Control

Sediment-control standards that coincide with the tolerances established for purposes of soil conservation have the distinct advantage that a farmer is in compliance if he follows a conservation plan approved by the SWCD. These plans include a safety factor in that they are generally designed to protect the most erodible portion of the field. Since field slope gradients are seldom uniform, the average soil loss for the entire field is usually less than that on the slope the plan is designed to protect.

Uniformly applied sediment-control standards based on average annual soil losses are perhaps the most feasible starting point, because of their simplicity and because knowledge of precisely how much upslope soil movement can be tolerated is inadequate. But if the initial standards fail to attain the desired level of water quality control, the next step should be a range in standards to suit the requirements of various local conditions rather than successive lowerings of uniform limits. Uniformly lowering soil loss limits to attain higher water-quality goals would unnecessarily remove substantial acreages from grain production.

Before quantifying gross-erosion limits for cropland, specific objectives of the limits should be defined. Uniform soil loss tolerances reduce the total quantity of sediment produced. This is important for reduction of direct damage by deposition on upslope areas, on flood plains, and in lakes or drainage ditches. But for control of water pollution from nonpoint sources, other aspects of the problem may be more important than the amount of soil eroded from a particular field slope. These include: upslope deposition, composition of the sediment, the protection needs, and fluctuations in sediment loads.

Upslope Deposition

For control of water pollution from nonpoint sources, soil material eroded from a field slope but deposited in terrace channels, field boundaries, or elsewhere along the path followed by the runoff enroute to the stream system is irrelevant. The fractions of sediments eroded from upslope areas that are delivered to a continuous stream system or lake range from less than 10% to nearly 100%. Uniform limits on erosion rates will allow a wide range in quantities of delivered sediments. Estimating sediment delivery ratios was discussed in the preceding section. Low sediment delivery ratios are of little relevance to preservation of the eroding cropland, but they are highly important for water quality control. Basing sediment standards on gross erosion minus the estimated upslope deposition would achieve more uniform control of sediment quantity and allow greater cropping flexibility. This would be a great improvement, but sediment quantity is not the only important criterion.

Composition Of The Sediment

Sediment traps or settling basins trap primarily the coarse material. Clay, fine silt, and light soil aggregates remain in suspension much longer than the coarse material and are the greatest concern as a source of turbidity and carrier of chemical compounds. There is some particle-size selectivity in erosion, but generally the composition of washoff material as it leaves the field is closely related to that of the soil from which it is derived. There is substantial size selectivity in the transport and deposition phases, but the composition of the sediment as it leaves the field will determine the proportion of fine material available for transport in suspension. Thus, for pollution control, variability in soil-loss limits should be related to soil texture.

Protection Needs

Sediment standards could also be selective in relation to the needs of the particular body of water being protected. For example, controls need to be more intensive for land draining into recreational waters and urban water supplies than for land draining into major river channels.

Fluctuations In Sediment Loads

Short-time high sediment yields are much more relevant for pollution control than for preservation of the land resource. The average annual soil loss from a particular crop system on a given field is the mean of yearly losses that may differ tenfold, or even a hundredfold, due to differences in the cover and management effects of the crops in the system, fluctuations in rainfall erosivity, and intermittent crop failures.

The soil loss equation shows that under conditions where a 6-year rotation of corn-corn-corn-wheat-meadow-meadow would average 5 tons of soil loss per acre per year with conventional planting and tillage, the first-year corn would average about 4 tons, second-year corn 9 tons, third-year corn 14 tons, wheat 2.7 tons, and meadow 0.2 ton. On the average, at least half of the soil loss from the corn would occur during the first month after preparation of the clean-tilled seedbed. Appendix tables in Agricultural Handbook No. 282 (*96*) show that about one year in ten the rainfall-erosivity factor is likely to exceed its local average value by 50 percent, and one year in twenty by 75 percent. If a 20-year rainfall occurred in the third corn year, the predicted soil loss for that year on this field would be 1.75 times 14 tons, or nearly 25 tons, even though the longtime crop-system average would not exceed the 5-ton limit.

Soil loss variability due to fluctuations in rainfall or occasional seeding failures cannot be prevented, and yearly or seasonal rainfall differences can be predicted only on a probability basis. Because these differences interact with other erosion factors, specific-storm soil losses can presently not be accurately predicted. However, for each crop in the system, the effects of fluctuations in rainfall tend to average-out over long time periods, and the differences in cover and management effects of the crops in a particular system are reasonably well known. Therefore, the average annual soil loss for each year in a crop sequence can be predicted by use of the Universal Soil Loss Equation with about the same accuracy as crop-system averages. This is done by deriving factor C on a yearly basis by the method illustrated in Agriculture Handbook No. 282.

Limits prescribed on a crop-year basis would reduce the frequency of very high single-year or single-event soil losses. In the preceding example, a 5-ton limit on the design loss in any year of the cropping system would require the use of good residue management for the second-year corn and no-till planting in shredded-cornstalk mulch for the third corn year. If the soil and climate were not compatible with no-till planting in residue cover, the third-year corn would need to be omitted from the cropping system.

However, crop-year soil loss limits would need to be higher or more flexible than the present rotation-average tolerances. If not, they would prevent production of corn, soybeans, or other rowcrops on numerous fields where these crops can be grown in rotation with meadow and small grain, and they could also eliminate the acceptability of periodic clean plowing for weed and pest control on fields that are usually no-till planted. The reason for this is that crop-year limits would allow no credit for much more drastic reductions in soil loss during other years in the crop system.

Modified Sediment Standards

A possible alternative would be to continue the present crop-system-average tolerances and superimpose limits on the maximum design loss for any one year in the system. The first limit would allow credit for meadows and other low-erosion crops in a system designed to preserve the productive capacity of the land. The second limit could be sufficiently higher to avoid unnecessary restrictions on land use and yet guard against frequent occurrence of very high single-year sediment yields.

The second limit would take into account such factors as the intermittent more erodible conditions that cannot be avoided, upslope deposition, soil texture, and specific control needs for the location. Upslope deposition could be increased by requiring use of sediment traps or filter strips of grass or small grain across the lower end of a field in the years when it is plowed. The same requirement could apply to the second and third corn years in sod-based rotations.

RESEARCH NEEDS

The past 40 years have brought great progress in erosion control, but serious erosion and sediment damages are still far too frequent. Population pressures, increased export demands for agricultural products, and more substitution of large machines for manpower changed the erosion problems and intensified the hazard on many millions of acres of productive cropland. Larger fields generally mean longer continuous slopes. Extensive monoculture sacrifices the potential residual effects of sod-based systems. Large equipment greatly increases production per man-hour but is not compatible with following the field contours on much of the cropland.

Furthermore, soil conservation and sediment control are two individual goals and do not have the same requirements. Personnel and resources available for erosion research in recent years have been insufficient to keep pace with the changing needs. Research is particularly needed in the following general areas. This research will involve many preliminary and secondary investigations that are not listed.

Sediment Delivery to Stream Systems

Average annual erosion rates on cropland can be predicted with reasonable accuracy, but the percentage of this eroded soil that reaches a continuous stream system cannot. Sediment delivery ratios as usually defined by geologists for dealing with downstream sediment problems are influenced too much by stream transport efficiency and sediment accretions from nonagricultural sources to provide the information needed for control of water pollution from nonpoint cropland sources. If the sediment delivery ratio is used for this purpose, it should be defined as the ratio of sediment delivered *at the place where the runoff water enters a continuous stream system* to the gross erosion from the drainage area above that point.

The sediment delivery ratio, by either definition, represents an attempt to reflect deposition and sediment accretions in a single factor. The net effect of the two processes would be difficult to relate to drainage-area parameters in a regression equation because large amounts of deposition and large sediment accretions can occur in the same drainage area and balance each other. Prediction equations for deposition and for sediment accretions from runoff-induced erosion below the field areas need to be separately derived, each as a function of the drainage-area parameters pertinent to that process. Such equations will facilitate estimating the effects of cropland erosion control not only on the amount of sediment delivered to the stream system but also on the composition of sediment yields farther downstream. Development of a better understanding of the basic sedimentation and erosion processes involved between the time when runoff leaves a field area and when it reaches a continuous stream system is one of the greatest erosion and sediment research needs.

Mathematical Modeling

Recent progress in development of mathematical models for simulating the erosion and sedimentation processes on field-size areas and on watersheds has demonstrated the potential of such models to predict temporal and spatial distribution of erosion and sedimentation and to predict specific events more accurately. However, some of the basic relationships assumed for these initial models need research testing, and the parameters need to be defined for a wide range of field conditions. Relationships describing erosion and deposition in channels and gullies also need to be derived.

These models are generally complex and difficult to use in the field. A relatively simple model that computes individual-storm soil losses more accurately than the Universal Soil Loss Equation is needed. Such a model can use the basic format of the USLE, but it will need separate erosivity factors for rill erosion and interrill erosion, and their relationships to the other factors in the equation will need to be determined. Use of volume and peak rate of runoff to predict rill erosion shows promise, but it requires derivation of a cropland-runoff prediction equation.

Improvement of the basic models, and research to determine the needed parameter relationships, should be emphasized. Such models can provide more dependable interpretation and extrapolation of field-plot data, and the predictions of spatial and temporal variations in erosion and deposition are needed for both conservation and pollution-control planning.

Residue Management

Residue management is one of the major tools for erosion control. In the densely populated countries, few residues are usually available for erosion-control use because they are needed for other purposes. We may soon have similar problems in this country if crop residues become economically profitable sources of energy, concentrated feeds, or building materials. Research must determine the optimum treatment and placement of very limited residues and the optimum amount and type of associated tillage required to minimize erosion in the absence of what we now consider adequate cover.

Neither has the optimum amount and placement of residues where they are abundantly available been determined. Optimum placement of a portion of the residue may permit incorporation of the remainder into the topsoil. This may reduce soil-temperature and wetness problems without decreasing the erosion control.

Critical Slope-Length Limits for Practice Effectiveness

Critical slope-length limits for effectiveness of partial mulch covers and favorable microtopographies provided by conservation tillage practices need to be determined.

If limits can be defined in terms of depth and velocity of runoff, they can then be related to soil, topography, and rainfall characteristics for guidance in field application. Clear definition of critical slope-length and drainage-area limits is needed for improved terrace spacing design and to prevent unexpected failures of some agronomic practices. The investigations should include evaluation of anchored versus loose mulches and of different types of residues.

Slope-length limits for effective contouring need to be more accurately defined in relation to permeability, soil stability, crop cover, and other factors. Successive row breakovers can result in rill erosion that more than offsets the reduction in sheet erosion effected by the contouring.

Erosion Index for Special Conditions

The EI parameter is a good indicator of the erosive potential of the rainfall and runoff in most of this country, but there are a few conditions for which further investigation of this factor is urgently needed. The erosivity of surface runoff that is not directly associated with drop impact needs to be evaluated, such as runoff from thaw and snowmelt. This item is particularly important in the Palouse Region of the Northwest. The effects of soil-surface shielding by ponded or very slowly moving runoff also need to be identified and evaluated. These effects may account for the difficulties experienced with the EI parameter on the Coastal Plains of the Southeast.

Topographic Factor

The topographic factor needs further research, both with reference to factor interactions and with reference to long or steep slopes. The effects of slope length and steepness on soil erosion are known to be more variable than indicated by existing formulas. There is evidence that they are significantly influenced by mutual interaction and by interactions with cover, soil texture, and rainstorm characteristics (or runoff rate). These interaction effects need to be quantified so that variations in the topographic factor can be predicted. This is important both for soil conservation planning and for pollution control guides.

Topographic effect also needs to be determined for steep roadbank and construction slopes and for long watershed slopes. Existing slope-length and steepness formulas were derived from data on slopes not steeper than 18% and, with only one exception, not longer than 270 feet. Extrapolation of the formulas to slopes that far exceed these dimensions is quite speculative.

Soil Erodibility

The soil-erodibility nomograph needs to be augmented for greater accuracy on high-clay subsoils and on sandy loams. Effects of soil chemistry on erodibility need further investigation and quantification. Susceptibilities of soils to sheet erosion and to rill erosion should be evaluated separately, and influences of montmorillinitic clays need further study. Stripmine areas and spoilbanks need specific research attention.

Particle size sorting in erosion and sedimentation is important for pollution control and has not been adequately investigated.

Runoff Equation

A cropland runoff equation designed for general field use would be a valuable asset in pollution control planning. It could also provide an additional factor for the Universal Soil Loss Equation that would improve its accuracy, particularly for moderate storms.

Sediment Traps

Vegetated filter strips, settling basins, and sediment traps can be used to cause sediments to deposit near the point of origin, but more needs to be learned regarding their design for optimum trapping efficiency and their particle size selectivity. Also, extreme runoff events may pick up substantial amounts of sediment from the traps. The probabilities of such occurrences and methods of minimizing them need to be investigated.

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CHAPTER 4

NUTRIENT ASPECTS OF POLLUTION FROM CROPLAND

M. H. Frere

Nutrients are naturally occurring chemicals essential for plant growth. Sixteen elements are essential for the growth and reproduction of most plants. Most soils are lacking in adequate amounts of nitrogen, phosphorus, and potassium. Hence, fertilizers containing these nutrients are essential to maintain the current level of agricultural production. The other nutrient elements may be added as impurities in the fertilizer or to treat specific nutritional problems. Present evidence indicates that nitrogen and phosphorus are the principal nutrient pollutants and, therefore, only these nutrients are considered in this chapter.

A major source of nutrients reaching water bodies in this country is sewage, both from municipal treatment plants and nonsewered residences. These are point sources of pollution and extensive efforts are underway to limit their contributions. Runoff from rural land is another major source. Unlike point sources, runoff integrates the contribution of nutrients and water from a wide variety of landscapes that are continuously changing with time. It must be recognized that nutrients leak from the system even when fertilizer is not applied and while we cannot eliminate nutrient losses, it is desirable to minimize them.

While a number of review papers have been written about nutrient losses (*47, 76, 89, 98, 145, 167*) the emphasis of this chapter is on the practices that can control nutrient losses and the background necessary to use these practices effectively. This chapter reviews the literature and summarizes the data for nitrogen and phosphorus leaving cropland by runoff, erosion, and leaching. The material covered is limited to precipitation-induced transport of the nutrients from cropland and improved pastures. Beyond the scope of this overview are the very important problems of irrigation return flow, wind erosion, and losses from waste disposal areas.

It must be emphasized at the beginning that the dynamic system under consideration is very complex. The wide variety of climates and landscapes provides such a wide range of results that there is no typical case. Complications are introduced by difficulties in chemical analysis for nutrients in water samples. Numerous procedures have been followed for chemical analysis. Changes in nutrient form can occur between the time the sample is taken and when it is analyzed. Some of the nutrient reported as soluble could have been associated with colloidal material not removed. The practical significance of these complications is unknown, but they are noted at this time to warn the reader of the limitations of the data associated with nutrient pollution.

THE PROBLEMS

Two problems are associated with nutrients in the aquatic environment: the water may be toxic to humans, animals, or fish when the concentration of certain nutrient forms exceeds a critical level; and eutrophication may be accelerated.

Toxicity

Case (*28*), Lee (*92*), and Winton (*181*) reviewed the problems associated with nitrates and nitrites in drinking water. The nitrite form of nitrogen, which is the most toxic, interacts with components in the blood to interfere with oxygen transport. Methemoglobinemia, the technical name given to this illness, is often called "the blue baby syndrome" because infants are very susceptible. Most of the problems with drinking water have been associated with farm wells with faulty well casings and located close to manure concentrations such as barnyards.

Nitrate is 5 to 10 times less toxic than nitrite and healthy mature animals with single stomachs are able to excrete nitrate in the urine. Cattle, young animals, and children convert some of the nitrate to nitrite in their stomachs and can develop methemoglobinemia. Since food also contains nitrite and nitrate, the response to nitrate in drinking water could be quite variable. The U. S. Public Health Service Drinking Water Standards of 1962 set the limit for nitrate at 10 mg N per liter (45 ppm nitrate). Armitage (*5*) reported the Recommended Drinking Water Standards of the World Health Organization to be: 0-50 ppm nitrate = recommended, 50-100 ppm nitrate = acceptable. There is some concern, however, that even nontoxic nitrate levels (chronic conditions) may lower resistance to environmental stresses and interfere with normal metabolism.

Dissolved ammonia is another form of nitrogen that can occur at levels toxic to fish. Microorganisms can generate free ammonia from organic matter in lake bottoms during summer stagnation periods (*164*). Trout are sensitive to 1-2 ppm ammonia (*35*) while goldfish appear to be less sensitive (*48*).

Eutrophication

Eutrophication is the enrichment of waters by nutrients and the ensuing luxuriant growth of plants. Much has been written about this subject in the last few years (*100, 111, 153, 169*). Rapid growth of algae is the greatest and most widespread eutrophication problem in most states (*2*). Algae can create obnoxious conditions in ponded waters, increase water treatment costs by clogging screens and requiring more chemicals, and cause serious taste and odor problems (*17*). When a large mass of algae dies and begins to decay, the oxygen dissolved in the water decreases and certain toxins are produced, both of which kill fish (*49*). The complexities of the ecosystem are illustrated by the observation that the nutritional status of a species of algae can vary from lake to lake or even from different areas and depths of the same lake on the same day (*39*). Streams, however, do not age in the same sense as lakes, but their biological productivity can be increased by added nutrients (*69*). For example, phosphate from farm land was a very beneficial and important factor in the high production of brook trout in Canadian streams (*137*).

Aquatic plants require a number of nutrients for growth, but nitrogen and phosphorus appear to be the ones accounting for most of the excessive growth. Sawyer (*128*) concluded that eutrophication becomes a problem when the concentration of inorganic nitrogen exceeds about 0.3 ppm and inorganic phosphorus exceeds about 0.015 ppm. These concentrations of the inorganic forms of nutrients are maintained by microbial conversion of organic forms so the total input of nitrogen and phosphorus per unit area of the lake (loading rate) is important (*57*). Current international guidelines for eutrophication control are 1.8 to 4.5 lbs. of P and 45 to 90 lbs. of N per surface acre of lake per year (*169*).

The various roles of nitrogen in eutrophication have been recently reviewed (*24, 53*). Aquatic organisms assimilate nitrate and ammonium. Ammonia and amino acids are excreted by live organisms and released by decaying organisms. Fungi and bacteria can convert the organic nitrogen in dead plant material and sediment to ammonia and nitrate. Whenever the environment becomes anaerobic, in the presence of decomposable organic matter, nitrates are denitrified to gaseous nitrogen compounds. Bouldin (*20*) estimates that the daily loss of the nitrate in the bottom sediments can be 7 to 15 percent by microbiological denitrification and 2 to 28 percent of the ammonia by volatilization.

Some additional inputs of nutrients are often overlooked, such as aquatic birds, leaves, dust, and pollen. Another source is the fixation of atmospheric nitrogen into organic nitrogen by a number of organisms such as the blue-green algae. This process is considered to be adaptive in that it occurs when other sources of nitrogen are depleted.

Kramer et al. (*86*) and Lee (*93*) reviewed the role of various phosphorus compounds in eutrophication. Soluble orthophosphate is usually regarded as completely available for algal growth. Soluble organic phosphates and polyphosphates are probably not too available, but are readily converted to orthophosphate. Finally, phosphate in particulate organic matter and adsorbed to mineral sediments is usually only slowly released. The adsorption capacity of the sediment for phosphate ranges from low for quartz sand to very high for certain silicate clays.

Sediment low in phosphate will usually remove phosphate from solution as it settles out (*63, 64, 86, 93, 179*). When the sediments have high P contents and the environment around the clay is electrochemically reduced, then some of the phosphate can be released to the solution (*86*). This released phosphate may form the mineral apatite, which is relatively insoluble (*179*), or if there is a mixing process, such as caused by wind, the phosphate is redistributed through the lake (*63, 178*).

Because a lake's ecological system is so complex, Shannon and Brezonik (*132*) devised an index of seven parameters to quantitatively characterize the trophic state of a lake. For 55 lakes in Florida, the relation between this index and the loading rates of nitrogen and phosphorus showed that an increased phosphate loading was statistically the more important. An additive form of the loadings accounted for 60 percent of the variability in the index.

SOURCES OF NUTRIENTS

Fertilizers are well known as a source of nutrients on cropland, but they are not the only source and sometimes are not the most important source. Fertile cropland soil is a pool of nutrients with different degrees of availability to the crop and to the transport processes of runoff, erosion, and leaching. Precipitation and animal wastes are other sources. In addition, legume crops, with the assistance of microorganisms, can biologically convert atmospheric nitrogen into organic nitrogen.

The relative importance of each of these sources depends on a number of factors, such as the geographical location with its relation to climate and soils and the crop management practices previously and presently used. Table 1 shows the estimated 1969 national nitrogen inputs (*119*) and estimates for several watersheds in Wisconsin (*13*). The national input for phosphorus in fertilizer is for 1973 (*79*), the manure input is from Table 3, and the inputs from plant residues and precipitation were calculated from the N inputs and N/P ratios of 7.5 (Table 17, Vol.I) and 100 (Fig. 3), respectively.

On the national level, fertilizer is a major input and manure is a relatively small input. The relative contribution of these sources can be reversed for a specific geographic location, as shown by the data for Wisconsin watersheds.

Nutrient Cycles

Nitrogen

Volumes have been written about nitrogen behavior. Two comprehensive reviews are: "Soil Nitrogen" (*10*) and "Soil Organic Matter and its Role in Crop Production" (*3*). Figure 1, adapted from Stevenson (*149*), illustrates the numerous compartments and pathways of nitrogen.

Most of the reactions in the soil portion of the cycle are microbial and thus the rates are sensitive to temperature and moisture. Warm (90° F) and moist (water in 80 percent of the voids) are optimum conditions for cycling within the soil. The conversion of organic nitrogen to nitrate (ammonification and nitrification) is often called mineralization. A study of 39 soils from across the United States showed that the rate of mineralization was proportional to the pool of mineralizable nitrogen. The size of this pool was not highly correlated with the total organic matter or total nitrogen (*148*). Thus, some forms of organic matter are readily converted to mineral forms whereas other organic forms are not. Part of the stable forms may exist as metal-organic and organic-clay complexes.

Soils contain 0.075 to 0.3 percent total nitrogen or 1,500 to 6,000 lbs. per acre in the top 6 inches (*5*). Soils

Table 1. Sources of nitrogen and phosphorus on a national and a watershed scale

Source	National				Wisconsin watersheds			
	Nitrogen		Phosphorus		Nitrogen		Phosphorus	
	Million tons	*Percent*	*Million tons*	*Percent*	*lbs/acre*	*Percent*	*lbs/acre*	*Percent*
Fertilizer	6.8	45.9	2.2	76	10	8.5	8	32
Fixation	3.0	20.3	0	0	12	10.3	0	0
Manure	1.0	6.8	0.4	14	42	35.9	12	48
Plant residues	2.5	16.9	0.3	10	45	38.5	5	20
Precipitation	1.5	10.1	0.01	0	8	6.8	0	0
Total	14.8		2.9		117		25	

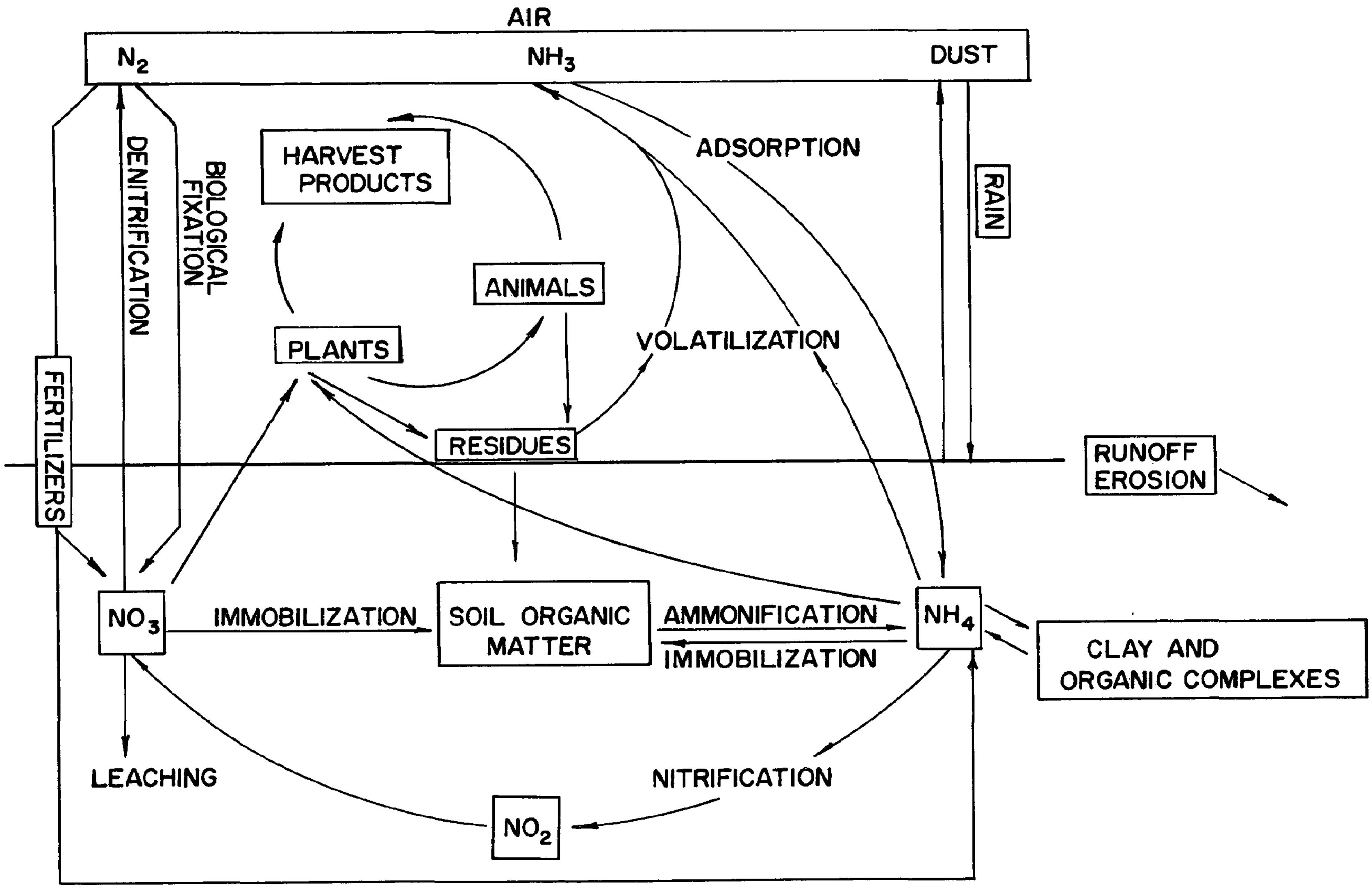

Figure 1.–The nitrogen cycle in agriculture.

cultivated for 100 years can still release 30 to 60 pounds of nitrogen per acre per year and have drainage waters with 5 to 10 ppm nitrate nitrogen. Fertile soils in the Corn Belt are estimated to release 120 lbs. of N per acre (*1*). In the semiarid west, some fields are fallowed (kept free of vegetation) every other year to accumulate moisture. The increased moisture also promotes mineralization of 20 to 50 lbs. of N/acre (*37*).

When ammonia (a gaseous form of nitrogen) reacts with the water it forms positively charged ammonium ions. These ions are held by the negative charge of the clays as exchangeable cations. Some of the ammonium ions can be trapped or fixed between clay platelets.

Ammonia absorption by the soil has not been considered in the past as a major path of nitrogen input. However, in areas of high ammonia concentrations, such as downwind of industries or feedlots, the soil, lakes, and plants have absorbed from 20 to 70 pounds of N/acre/yr (*55, 67, 68, 138*).

Immobilization, the reverse of mineralization, is the part of the N cycle that converts nitrate and ammonium into organic forms. It occurs under aerobic or anaerobic conditions and basically involves the uptake of the mineral forms by microorganisms in the synthesis of cell tissue. Whenever organic residues low in nitrogen are being decomposed, mineral nitrogen must be used because the carbon-to-nitrogen ratio of microbial tissue is on the order of 5-10:1. Dead microbial tissue then becomes part of the organic matter pool that can be mineralized. A major problem in quantifying the immobilization process is that it is impossible, without nitrogen tracers, to measure the small amount of the product in the large organic-matter pool.

Denitrification is not well understood but appears to be a very important part of the nitrogen cycle affecting environmental quality. Denitrification is the use of nitrate by anaerobic microbes for oxygen and results in the production of nitrogen and nitrogen oxide gases. The necessary anaerobic conditions are most prevalent when the water content of the soil is high, which is the same condition needed for leaching. Another requirement is a supply of carbon for an energy source. Lack of useable carbon may be the factor that prevents all the nitrate in the leachate from being denitrified. Carbon is usually not very mobile and, therefore, once the nitrate passes below the root zone, the opportunity for denitrification is limited (*114*). Complete waterlogging is not essential for denitrification. Since the soil contains a wide range of pore sizes, an unsaturated soil can have areas where the water contents and microbial activity are sufficient to produce an anaerobic environment and denitrification. Quantification of this process has been limited. Most of the estimates have been based on nitrogen budgets where all unaccounted-for nitrogen is assigned to denitrification. The nitrogen gases produced are very difficult to measure under field conditions.

Average losses have been estimated at 10 to 30 percent of the total yearly mineral nitrogen input (*26*). When excessive rates of nitrogen are applied, as much as 50 percent can be lost (*99*).

The inputs of fertilizer, biological fixation, animal wastes, and precipitation; and the losses by leaching, runoff and erosion, and plant uptake (avoiding excessive fertilizer use) will be covered in subsequent sections.

Phosphorus

The phosphorus cycle shown in Figure 2 is a lot less complicated than the nitrogen cycle, although it has some of the same paths. Before considering the similar paths, we will examine those reactions in the soil which are unique for phosphorus. Black (*15*), Olsen and Flowerday (*115*) and Ryden et al. (*125*) have prepared comprehensive reviews of the subject.

The phosphate concentration in the soil solution is low, usually 0.01 to 0.1 ppm P, although the total P in the soil ranges from 100 to 1,300 ppm (*13*). The mineral forms of phosphorus–calcium, iron, and aluminum phosphates–have very low solubilities and the phosphate is highly adsorbed to clay minerals. The organic forms of phosphate have not been studied as extensively as have the inorganic forms. However, the organic part of the total phosphorus can range from 3 to 75 percent.

Because of the low solution concentrations and high degree of adsorption, phosphate tends not to leach. After 286 lbs. of P had been applied in 11 years, the plant "available" level had increased only 18 lbs. and very little had moved below 12 inches (*32*). After 82 years of fertilization, the total P had been doubled, but no added P was found below 54 inches (*16*). The availability of phosphate in the soil decreases exponentially with time. However, soils vary greatly in their conversion of added P to insoluble forms (*90*). Recent work (*52*) indicates that chemical reactions immobilize more than 50 percent of added soluble phosphate in a few hours and an additional 10 percent in a month or so. The amount of biological immobilization into organic phosphates that occurs simultaneously with the chemical reactions depends upon the amount of biological activity.

Precipitation

The amounts of nitrogen and phosphorus added in precipitation are generally low and, for cropland, they

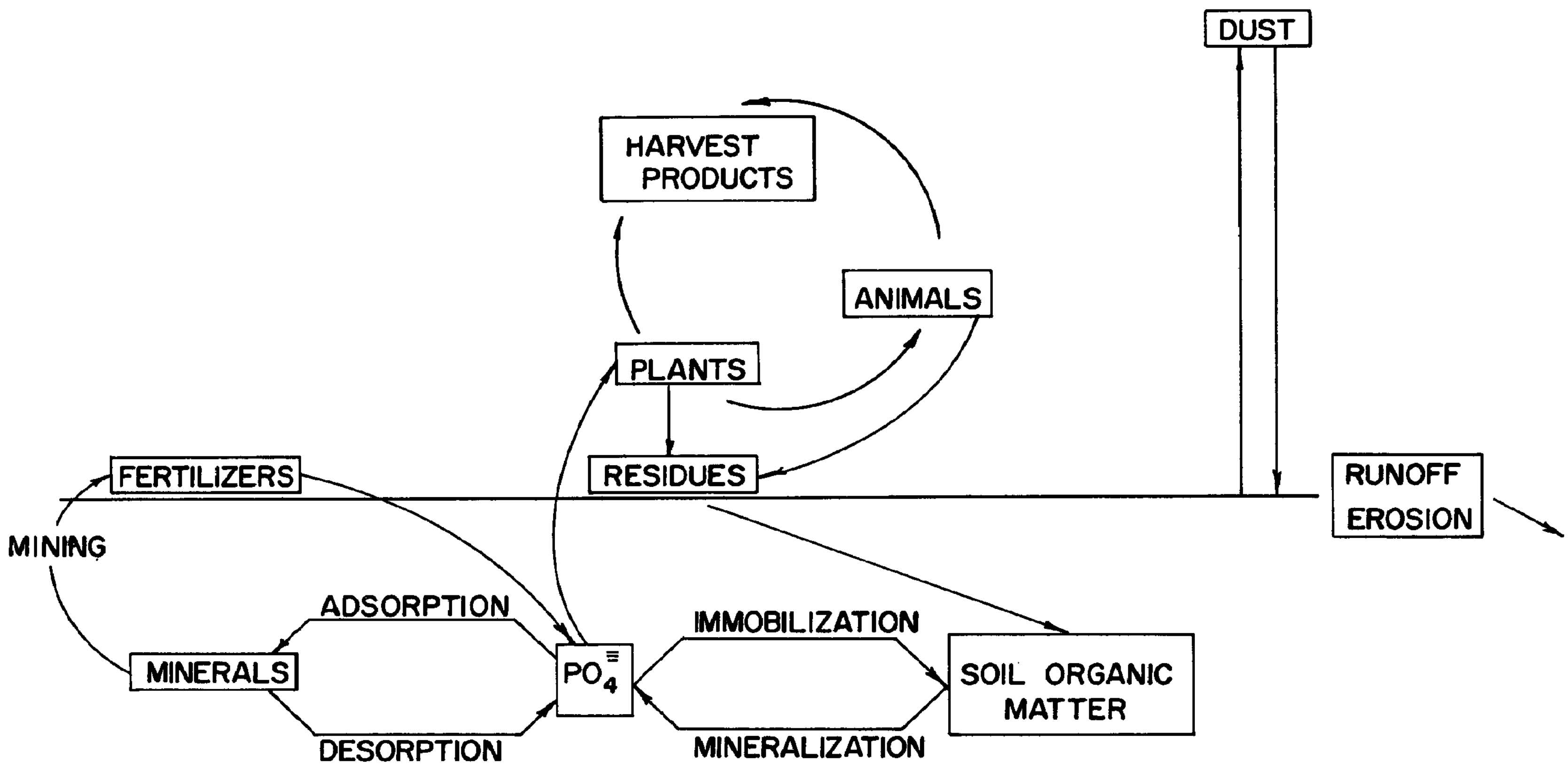

Figure 2.–The phosphorus cycle in agriculture.

are negligible in comparison to other inputs. On forests, unimproved pasture and rangeland, and lakes, the input of nutrients added by precipitation can be significant. The concentrations of nitrogen and phosphorus in rain and snow vary not only across the country, but within short distances and during a storm.

Uttormark et al. (*166*) list 40 factors that can influence the concentration of nutrients in precipitation. Feedlots, industrial urban centers, power plants, disposal sites, etc., are all relatively local sources that can increase the general regional level of nutrients. They prepared a map of the United States delineating areas of different nitrogen contribution in rainfall. The Lake States had the highest contribution, about 2 to 3 lbs. of N/acre/yr., whereas the Western States had less than 1 lb. of N/acre/yr. Dry fallout was not included.

Seasonal maps of ammonium and nitrate in rain across the United States (*73*) show that the highest concentrations are in the spring and summer. It appeared that the rainfall concentrations might be related to the soil because the Southeast acidic soils are the lowest. Ammonium concentrations may change as much as 10-fold during the year, but nitrate changes much less.

When rainfall samples are taken at different times during a storm, the nitrogen concentrations are often slightly higher during the first part (*24, 50*). This could be due to evaporation from the drops into dry air or the washout of dust.

The phosphate input in most places may be associated with dust, either as dry fallout between storms or washed out of the atmosphere with rain. Dust or dry fallout is a very important component associated with the precipitation input of nutrients. It has been estimated (*166*) that in arid regions 70 percent of the nitrogen in uncovered rain gages is from dust. Storms blowing in from oceans are quite low in phosphorus. Phosphorus can also be associated with ash and smoke, as indicated in concentrations of 0.24 ppm phosphate in rain at Cincinnati compared to threefold less at a rural location near Coshocton, Ohio (*174*). A yearly input of 0.2 to 0.6 lb. P/acre/yr has been estimated (*24, 110*). One problem associated with evaluating the importance of dust is the absence of a measure of the loss by dust into the atmosphere. There is no evidence that snow has any different concentrations of nutrients than rain would at the same time of year.

Fertilizers

Fertilizer use to improve crop yields dates from antiquity. Sanskrit writings of 3,000 years ago note the value of dung for fertilizer (*103*). Guano and Chilean nitrate were available in Europe as a fertilizer over 100 years ago. Yields from field tests started at England's Rothamsted Experiment Station in 1840 have been maintained near maximum by the use of manure and fertilizer while those from unfertilized plots have decreased to an uneconomic level.

Organic forms of nitrogen were the cheapest sources of nitrogen until about 1900 (*70*). Before the 1950's, sodium nitrate and ammonium sulfate were the principal nitrogen sources. Then ammonium nitrate became the leading source, only to be surpassed by anhydrous ammonia and urea in the 1960's. Phosphate fertilizers also changed in the 1950's from normal superphosphate to concentrated superphosphates. These are trends towards the use of more concentrated forms, thus reducing shipping charges per pound of nutrient. The source of the nutrient makes no difference to the plant because of the extensive cycling in the soil. Some sources, such as ammonium sulfate, can increase the soil acidity if used for extended periods.

Table 2 shows how fertilizer use on four major crops has changed over a 10-year period. The acreage of corn, wheat, and cotton harvested has remained relatively constant except in 1974 when acreage controls were removed. The acreage of soybeans has steadily increased. Except for cotton, the percentage of acres fertilized has consistently increased over the years. Note also that the yield per acre has tended to increase. Fertilizer rate has increased except for nitrogen in 1974 when short supplies and increased costs because of the energy crisis caused many farmers to reduce their application rate. The relative plateau of fertilizer use on cotton deserves some comment. Cotton is relatively sensitive to nitrogen and excess nitrogen can reduce yields. Cotton has been fertilized intensively for a long time and the optimum rates have evidently been found. Pests, such as insects, weeds, and disease, are probably limiting yield more than fertility.

Speculation on future fertilizer use is fraught with uncertainties. The problem is basically economics. How much fertilizer should the farmer apply to maximize his profits? The yield response to fertilizer is less for each subsequent increment of fertilizer and as the yield increases some other costs also increase. In the 1960's, farmers received $2.50 for each dollar invested in fertilizer and so they substituted an investment in fertilizer for additional land or labor (*62*). Real estate costs and wages increased over 250 percent since 1950 while plant nutrient costs decreased (*38*). While the energy crisis will tend to increase fertilizer costs faster than other costs, it will probably be sometime before they reach the level of other farm costs. Thus, farmers will probably continue to fertilize, but will carefully appraise the size of the applications and eliminate

Table 2. The change in fertilizer use on four crops in the past 10 years

Crop	Year	Acres harvested[1]	Area fertilized[2]		Fertilizer rate[2]		Yield[1]
			N	P	N	P	
		Millions	*Percent*		*lbs./ac.*		*bu./ac.*
Corn	1964	55.4	82	75	45	18	63
	1966	56.9	91	85	83	24	73
	1968	55.9	92	88	102	28	80
	1970	57.2	94	90	112	31	72
	1972	57.4	96	90	115	29	97
	1974	63.7	94	87	103	27	–
Wheat	1964	49.8	47	36	28	12	26
	1966	49.9	49	38	32	14	26
	1968	55.3	56	43	37	14	28
	1970	44.1	61	44	39	13	31
	1972	47.3	62	44	46	16	33
	1974	64.1	66	46	46	17	–
Soybeans	1964	30.8	6	10	13	11	23
	1966	36.5	17	24	14	15	25
	1968	41.1	21	27	12	17	27
	1970	42.1	21	27	14	16	27
	1972	45.7	22	29	14	18	28
	1974	52.5	22	28	15	18	–
							lbs./ac.
Cotton	1964	14.1	75	56	69	21	517
	1966	9.6	75	58	77	23	480
	1968	10.2	73	55	71	23	516
	1970	11.2	72	48	75	24	438
	1972	13.0	77	55	75	24	507
	1974	13.1	79	58	78	23	–

[1] United States Department of Agriculture, "Agricultural Statistics", Years 1964 to 1974.
[2] 1964-1970 data from "Cropping Practices" SRS-17, Statistical Reporting Service, USDA.
1972-1974 data from "Fertilizer Situation", FS-5, Economic Research Service, USDA.

excessive use. A food crisis that increases the price of farm products relative to fertilizer costs would stimulate fertilizer use. Fertilizer use in the United States will increase 5 percent per year because of rising populations, improved diets, and increased exports, according to a recent estimate (*78*).

Animal Wastes

Before commercial fertilizers came into common use, animal manure supplied most of the nutrients added to the soil. In the 1960's it was more expensive to load, haul manure several miles, and spread it than it was to purchase and apply commercial fertilizer. In addition to the economics, the convenience of commercial fertilizer caused animal wastes to become a disposal problem rather than a nutrient source.

The following discussion is based on information from several references (*29, 30, 96, 170, 180*).

Manure is the excrement of animals that contains the undigested food and the urine. The nutrient content of manure is different for different animals, type of feed, and amount of water consumed. In addition, the amount of bedding used to absorb the urine or presence of superphosphate to react with the ammonia, the method of storage, and the duration of storage influence the nutrient content of manure being applied to the soil.

Fresh manure contains 50 to 90 percent moisture, 0.2 to 6 percent total nitrogen, and 0.06 to 2.5 percent total phosphorus, on a dry weight basis. The old rule of thumb was that the typical ton of moist cow manure contained about 10 lbs. of total nitrogen and 1 lb. of total phosphorus when it was applied to the field. Present methods of confinement, feeding, and manure handling vary enough that this rule of thumb is no longer adequate.

Yeck et al (*183*) recently estimated that 5.8 million tons of nitrogen are excreted annually by livestock in the United States. The percentage that can be collected varies from zero for cattle on the range to nearly 100 for caged poultry. The work of a number of investigators indicates that about half the nitrogen collected is lost during storage, handling, and spreading before it can be incorporated into the soil. Thus, of the 2.4 million tons of nitrogen that they estimate can be collected, only 1.2 million tons are available as a substitute for fertilizer and a large part of this is already being applied.

The pollution potential isn't a result of animals per se, but of the manure they produce. Since there are maps showing the geographical distribution of animals (Figs. 20-24, Vol. I), we needed a method for conversion to manure and its nitrogen and phosphorus contents. From the collectible nitrogen previously described for each class of livestock, a 50 percent loss before incorporation was assumed. This can then be geographically distributed with the animals. To calculate the amount of manure associated with nitrogen one must assume some percentage of N in the manure being spread. The amount of phosphorus can likewise be calculated from the percent phosphorus. These percentages of N and P in the manure were estimated for the different kinds of animals from data of various researchers. These calculations are summarized in Table 3.

Not all of the nitrogen in manure is immediately available. Most of the nitrogen left in manure when it is incorporated into the soil is in organic compounds. Like soil organic matter, microbes must mineralize the organic nitrogen into ammonium and nitrate, which are taken up by the plant. About 40 or 50 percent of the organic nitrogen is mineralized during the first cropping season (*172*). Additional amounts are released in subsequent years so that yearly applications can build up the nitrogen-supplying capacity of the soil. Of course this "slow release" of nitrogen also means that some of it can be released when plants are not rapidly growing (fall and spring) and thus be available for leaching.

Biological Fixation of Nitrogen

One of the most recent reviews of the biological fixation of nitrogen is by Allison (*3*). Another good source of information is three chapters in "Soil Nitrogen" (*10*). Nonsymbiotic (free-living) organisms probably fix about 10 lbs. of N per acre annually. This amount is of little practical importance in the nitrogen balance of a cultivated field, but since it is comparable to the precipitation contribution, it is important in nonfertilized grass sods.

Nitrogen-fixing bacteria in a symbiotic relation with roots of certain plants, principally legumes, can fix sufficient nitrogen to support a grass-legume pasture. With low soil nitrogen, the amount of nitrogen fixed in effective legume nodules correlates closely with the dry weight of the legume tissue produced. Generally, the fixed nitrogen is produced only as the plant needs it and, therefore, plants with poor growth will not fix much nitrogen. Similarly, high soil nitrogen levels, such as from fertilization, will reduce the amount of nitrogen fixed because it is not needed for plant growth. There is some indication (*12*) that fresh organic matter from previously fertilized crops stimulates N fixation. Biggar and Corey (*13*) report 200 lbs of N per acre per year as being fixed in legume systems. Allison (*3*) cites several references where 150 to 300 lbs. are common and as much as 600 lbs. of N/acre/year was observed. He concludes that a grass-legume mixture may fix more N per acre than legumes alone because the grass will continually remove any nitrogen mineralized from the soil or plant material.

Table 3. Estimates of nitrogen and phosphorus in manure that is available for application to cropland

Animals	Vol. I. Figure No.	Collectible N[1]	Available N[2]	N content[3] in manure	Available[4] manure	P content[3] in manure	Available P[5]
		Thousand tons	*Thousand tons*	*Percent*	*Million tons*	*Percent*	*Thousand tons*
Beef cattle	21	650	330	2.5	13	0.8	100
Dairy cattle	22	670	330	2.0	17	0.6	100
Swine	23	600	300	2.8	11	1.0	110
Laying hens	24	250	125	4.5	2.8	1.7	50
Broilers	25	240	120	3.8	3.2	1.3	40
Totals		2,410	1,205		47.0		400

[1] Estimated amounts of nitrogen that can be collected (183).
[2] Available after application assuming 50% losses during handling, storage, and spreading.
[3] Estimated contents based on data from various authors.
[4] Calculated from available N and N content.
[5] Calculated from available manure and P content.

TRANSPORT FROM CROPLAND

Water pollution by nutrients from cropland involves one of three transport processes: leaching, runoff, and erosion. In nature the results of these processes are not easily distinguishable. Water may infiltrate into the soil and thus cause leaching, but a few feet or yards downslope the water with its dissolved nutrients may come to the surface and join the overland flow or runoff. Similarly, the distinction between runoff and erosion may be quite difficult when nutrients in the runoff are adsorbed on or released from the eroded sediment that the runoff water is carrying. The larger the drainage area of the stream sampled, the more mixed are the three transport processes. Some cases where these processes operated independently will be examined so that the system can be better understood.

The usual range of values is presented in Figures 3 and 4. Occasionally, more extreme values will be observed because the system is highly variable. Concentrations are not provided for sediment transport because the concentrations of sediment vary so much. Soils contain 0.075 to 0.3 percent nitrogen and 0.01 to 0.13 percent phosphorus and deposition of coarse material during transport can increase the concentrations in the transported sediment by 2- to 6-fold.

Leaching

Leaching is the process whereby soluble chemicals are dissolved and removed from the soil in water that is percolating through the soil. Nitrate is the principal nutrient form found in drainage waters because it is seldom adsorbed to the soil minerals. Some red subsoils in the southeast are an exception (*156*). Organic forms of nitrogen and phosphorus and the orthophosphate ion are seldom found to any extent in drainage water because they are held by the soil.

Three principal ways of evaluating leaching are by lysimeter studies, monitoring tile drains, and taking core samples. Each method has some limitations and it is important to recognize these limitations when interpreting the data.

Lysimeters

Drainage lysimeters are columns of soil in the field isolated by impermeable cylinders with facilities for collecting the water that drains out of them. Cylinder walls at or above the surface can prevent or reduce runoff and, therefore, increase the amount of leaching. Also, if suction is not applied, the bottom of the column must be saturated before drainage will occur. This saturated zone can provide an opportunity for denitrification and dilution. Some lysimeters do not contain undisturbed profiles while others are very shallow. Both conditions limit their usefulness for field interpretation.

Lysimeters have provided some valuable information about leaching. Kolenbrander (*85*) found that as the clay content of the soil increased from 10 to 50 percent, the nitrogen loss decreased from 40 lbs. N/acre/year at 18 ppm to 4 lbs. N/acre/year at 2 ppm. Wild (*177*) found that nitrogen mineralized in the soil didn't leach quickly in fine soil with cracks, but Kissel et al. (*80*) found that applied fertilizer, simulated by chloride, did move quickly through the cracks of a clay soil. Kolenbrander (*85*) reported phosphorus losses of 0.2 lb. P/acre/year at 0.08 ppm for both cropped and grass lysimeters, with and without fertilizer.

Tile Drains

Tile drains have recently provided most of the measurements of nutrient leaching. While tile drains sample a much larger area and thus provide a more integrated value, they may not accurately reflect the nutrient content and water volume leaving the field. One problem is to define the boundaries of the drainage field so that losses can be calculated on an area basis. Another is that the tiles short-circuit the drain paths with aerated conditions reducing the time and opportunity for denitrification (*157*). As an illustration of this condition, Thomas and Barfield (*157*) monitored an area where one-third of the water flow was from tile lines with nitrate levels of 15 ppm N while the rest of the seepage had only 3 ppm N. At lower flows, the seepage accounted for nearly 90 percent of the flow with nearly zero nitrate while tile lines had 10 percent of the flow with concentrations of 9 ppm N.

Many studies have been reported concerning the concentration and the average annual loss per unit area of nutrients in drainage waters (*83, 96, 125, 166*). The first feature to be recognized is the extreme variability in the data, both the loss per unit area and the concentration. Thus, an average value has little utility. One of the major factors causing the variability in the loss data is that the water flow, and thus the load of nutrients transported, varies greatly between dry years and wet years. The type of crop grown is another major factor. Nutrient concentrations are consistently lower in the drain water from grasslands and woodlands than in that from cropland. Grasslands and woodlands have a longer

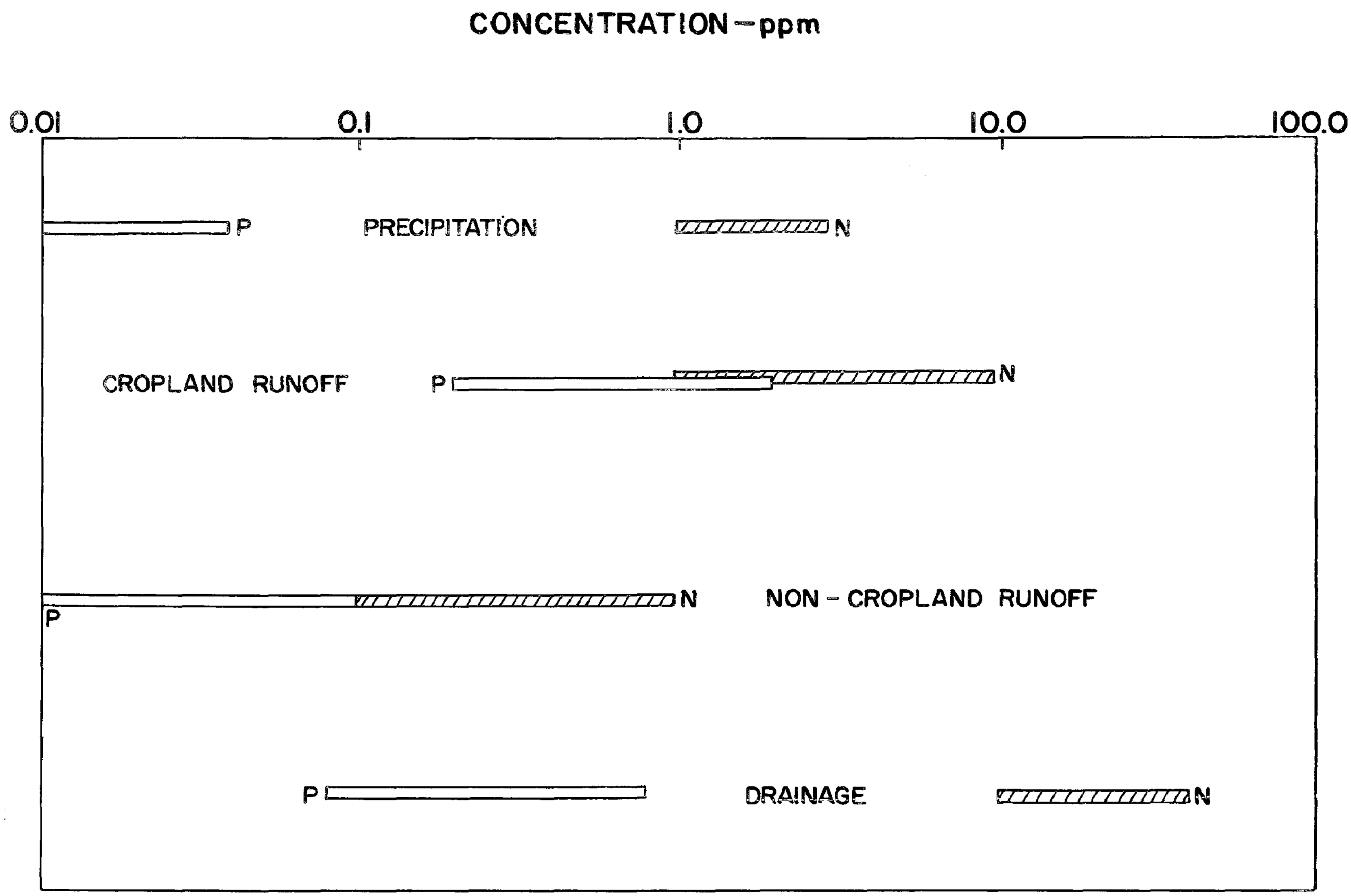

Figure 3.–Range of nitrogen and phosphorus concentrations in different waters.

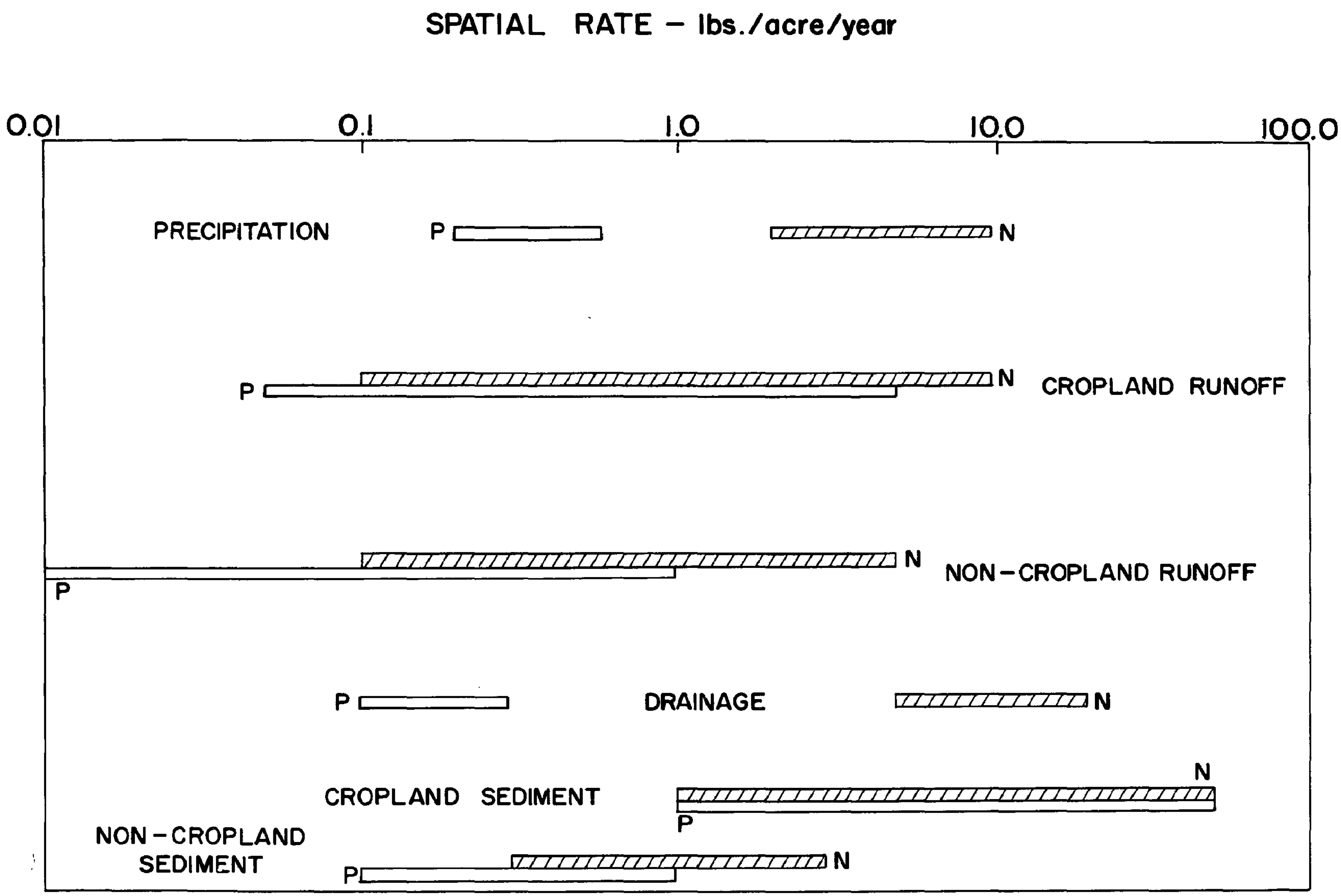

Figure 4.—Range of spatial rates of nitrogen and phosphorus in waters and sediments.

growing season in which to remove nutrients and reduce water flow. Also, even though they are located on soils of lower fertility, these lands are seldom fertilized because the economic return is insufficient. Generally then, the fertile land that is cropped will have the highest concentrations in the drain water and the most drain water. Fertilizer applied to these lands tends to increase the concentration of nitrogen in the drain water, but seldom that of phosphorus.

Soil Cores

Cores of soil can be taken from plots and fields, then separated into segments and analyzed. This provides a means to follow the movement of the nutrients down through the soil profile. In most soils, the distance the band of applied nitrate moves down through the soil is proportional to the amount of water in excess of that required to raise the water content to field capacity (*84, 94*). Water in excess of crop needs and soil properties such as texture and structure are important factors in determining the rate of leaching. In Wisconsin, Olsen et al. (*114*) estimated the annual percolation below the root zone to be 6 inches. A silt loam soil has a water capacity of 30 to 50 percent, thus nitrate could move 12 to 18 inches per year under these conditions and it would take at least 20 years to reach a water table at 30 feet.

In the semiarid Great Plains, leaching is not a hazard except when additional water is added by irrigation or fallow land prevents normal transpiration by plants. North Dakota soils wetted to 6 feet only occasionally in 40 years (*121*), but in Colorado, nitrate accumulated below the root zone in a wheat-fallow system (*150*). The average nitrate content in a 20-foot profile was 260 lbs N/acre for cultivated dryland compared to 90 lbs N/acre in native grass. Since very little fertilizer is applied to dryland crops in eastern Colorado, the difference between these averages is probably a result of nitrate production and leaching under cultivation and fallowing.

Runoff

Runoff occurs when the rate of precipitation exceeds the rate of infiltration. A heavy mulch apparently acts as a sponge to hold water as do the numerous small surface depressions. Surface-applied fertilizers can dissolve into this water held in the depressions and by the mulch. When this water becomes runoff it carries a load of nutrients. This is why the highest concentrations in runoff occur when there is runoff soon after surface fertilizer application. The longer the time between application and runoff, the greater is the chance for light rains to leach the fertilizer into the soil and even for water from the moist soil to dissolve the fertilizer so that it can be leached into the soil.

As a result of the factors just discussed, it is not unexpected that the concentration of nutrients in runoff varies from field to field and from storm to storm. The nutrient concentrations in runoff from small watersheds and fields remain relatively constant during a storm (*41, 129*) but these concentrations are usually higher than the base flow of streams in larger watersheds and thus the impact of storm runoff can be detected in streamflow (*51*).

Bradford (*23*) observed that the total N in runoff from fertilized plots was 60 percent higher when the rain occurred 3 days after application than when it occurred 3 months later. Rogers (*123*) reported 9 percent of the fertilizer was lost in the first rain and 4 percent in the second rain after phosphate was applied to pasture. On a watershed basis, Kilmer et al. (*79*) found less than 0.5 percent of the phosphate fertilizer was lost in a storm right after application. Dunigan et al. (*36*) found that more fertilizer was lost from top dressing (surface application) in 1972 with less rain than in 1973, because the first few rains in 1973 did not produce runoff but moved the fertilizer into the soil.

Nitrate, being very soluble, is usually leached into the soil by infiltration during the first part of the storm. Thus, the more infiltration there is before runoff, the lower the nitrate content of the runoff water. However, if the infiltrating water moves laterally and returns to the surface (interflow), its nitrate load is added to the overland flow. This return flow is more likely to occur as the size of the watershed above the sampling point increases. Barnett et al. (*8*) observed this phenomenon with one of three soils in Puerto Rico. On two soils, the runoff contained 0.3 percent of the fertilizer, but on the third, where most of the water infiltrated and then moved laterally through the soil, 7 percent of the applied fertilizer was lost. Sievers et al. (*133*) observed that most of the nitrate leached into a silt loam and a sandy loam soil gave a runoff concentration of 5 to 7 ppm N. Conversely, a silty clay loam soil with poor drainage produced more runoff with a concentration of 44 ppm N.

Usually the most fertile lands are cropped and, therefore, it should be expected that the water leaving cropland contains more nutrients than water leaving land in other uses. Losses from pastures, rangelands, and woodlands are usually much lower because the natural fertility of the soils is usually lower, fertilizer is applied less often if at all, and the amount of runoff is often less because there is more plant cover more of the time. Overgrazed lands have higher nutrient losses because of increased runoff and erosion. Concentrations of nutrients will also be high occasionally if animals have direct access to a stream, if animals are fed near streams, or if fertilizers and manures are surface applied. Phosphate concentrations in runoff may be higher from grasslands than from adjacent cropland after harvest because freezing and drying cause a release of nutrients from the vegetation (*163, 176*). Timber harvesting also causes a sudden release of nutrients to runoff waters from the decay of the trimmed foliage (*18*). The reduced water consumption after harvest also increases the annual nutrient loss. Uttormark et al. (*166*) and Brezonik (*24*) provide a summary of losses from land in various uses. Ryden et al. (*125*) provide a recent review of phosphorus losses.

Sediment

Sediment is the major transport vehicle for phosphorus and organic nitrogen. Raindrop splash and overland flow of water detach soil particles containing adsorbed phosphorus and associated organic matter. The flowing water transports the particles off the field. Transport capacity depends primarily on the volume and velocity of water flow. Whenever the velocity is reduced, such as by a flatter slope, the transport capacity is reduced and any sediment in excess of the reduced capacity settles out. Since the larger and heavier particles settle out first, the remaining sediment contains a larger percentage of the finer particles. The finer particles have a higher capacity per unit of sediment to adsorb phosphorus and, also, organic matter is lighter and tends to be associated with the fine particles. Thus, the transported sediment is richer in phosphorus and nitrogen than the original soil (*59*).

Bedell et al. (*11*) found that 98 percent of the sediment samples from 2- to 4-acre watersheds contained as much or more organic matter, phosphorus, and nitrogen than the soil. The degree of enrichment has often been described by an enrichment ratio, the ratio of the nutrient concentration in the sediment to its concentration in the soil of the watershed. Recently, enrichment ratios of 2 to 6 have been found for phosphorus (*130, 155*). Barrows and Kilmer (*9*) reported an average enrichment ratio of 2.7 for nitrogen and 3.4 for "available" phosphorus. Available means that fraction by chemical extraction that is expected to be available to plants.

Massey et al. (*104, 105*) found that the enrichment ratio was inversely related to the concentration of sediment in the runoff and the total amount of sediment lost. Doty and Carter (*34*) found that when the sediment concentration was highest at peak flow, the chemical and physical composition of the sediment was similar to that of the soil. At lower flows, the sediment concentration was lower, the percentage of clay in the sediment increased, and the enrichment ratio increased.

The loss of total N and P in sediment from cropland ranges from 1 to 50 or 100 lbs. per acre per year. Two factors can contribute to the high loss from cropland: the soil is fertile and contains a lot of nutrients per pound of soil, and tillage operations often leave the soil very susceptible in the most erosive part of the year. Noncropland areas often have a lower nutrient content than the cropland and also lower erosion rates because of more vegetative cover. An exception is the large amounts of sediment produced by gullies that are often prevalent in noncropland.

Nutrient Losses from Large Watersheds

As indicated at the beginning of this section, the nutrient composition of a stream reflects a mixture of the three basic transport mechanisms. The composition of streams changes with amount of flow, season of the year, and distance down the stream as new material is added from tributaries, seepage, and outfalls.

The Environmental Protection Agency (*165*) is undertaking a national eutrophication study, but at present only a preliminary analysis is available. The data indicate less variability in total nitrogen loss than in total phosphorus loss, and increases in the concentrations of nutrients in the water are correlated with increased density of animals in the watershed. A recent summary of the spatial loss for streams (*166*) reports a range of 1 to 13 lbs. N/acre/year and 0.03 to 2 lbs. P/acre/year, with averages of 5 and 0.4. Both studies indicate that forest land yields less nutrients than agricultural land and that the Midwestern states have higher losses than other states.

Most perennial streams have a number of outfalls for municipal and industrial wastes, which complicates any budgeting of sources. For example, agricultural land is estimated to have contributed all of the inorganic N, 49 percent of the total N, and 13 percent of the total P to the Potomac River in 1966 (*71*). The figures for the agricultural contribution to the Hudson River were 37 percent of the total N and 27 percent of the total P.

Streams are dynamic systems with living organisms assimilating the nutrients and particulate matter adsorbing or releasing the nutrients. Some of the nutrients removed from solution may go undetected in the debris moving along the bottom of the stream or floating on the surface. Phosphate-deficient sediments will remove phosphate from the solution (*154*). Thus, the concentration of phosphate in a stream changes as soil is added from noncropland and stream banks (*88, 173*) or gullies (*130*). Keup (*75*) provides a good discussion of the behavior of phosphorus in flowing streams. He cites studies on two rivers where the overall loss from solution was logarithmically related to the distance of streamflow below a point source of phosphorus. The coefficient was 0.01 to 0.02 per mile; that is, 14 to 30 miles for a 25 percent reduction.

EFFECT OF CONTROL PRACTICES

There is every reason to believe that nutrient loss from cropland can be controlled at an acceptable level if proper management practices are used. As pointed out by Klingebiel (*82*), soil surveys are an important basis for planning the optimum use of each field. By intensive use of fields with high crop production potential and low hazards from runoff, erosion, and leaching, the use of marginal land with higher hazards can often be reduced.

The effectiveness of management practices for the control of nutrient losses has not been quantitatively evaluated to the same degree as have the impacts on runoff and erosion. Only in recent years have adequate data on nutrients been collected.

The possibility of creating another problem by solving one problem should be the concern of all who make recommendations. The nitrate contamination of ground water in Runnels County, Texas (*87*) can be used as an example of this possibility. This land, which had been dryland farmed since 1900, had nitrate formed and leached below the root zone but not down to the water table. Extensive terracing after the drought in the early 1950's increased water retention and leached the nitrate on down to the water table. Thus, a nitrate leaching problem was created by the terracing done to solve a problem of limited moisture. While hindsight is much clearer than foresight, we should learn from previous experiences and examine our recommendations for secondary effects.

Erosion and Runoff Control Practices

Sediment is a major pollutant in itself. That it also carries nutrients and pesticides means that the first goal in controlling pollution from cropland should be to control erosion. For example, more than 97 percent of the N and P lost from some watersheds was associated with sediment lost primarily in the 2 months after planting (*27*).

Soil conservation practices have been stressed for over 40 years and sufficient data have been collected to permit fair predictions of average annual soil loss. Old principles are continually being used to create new methods of control (*140*). Data on the effectiveness of these practices for controlling nutrient losses are adequate for only qualitative predictions. Generally, a reduction in sediment loss provides less of a reduction in nutrient loss.

Residue Management

The greater the amount of residue left on a field, the greater the reduction in erosion. Zwerman et al. (*184*) reported that leaving the crop residues instead of removing them decreased runoff 50 percent and did not change the nutrient content of the runoff water. Thus, losses of nutrients in runoff were reduced 50 percent. Losses of nutrients with sediment were also reduced. Romkens et al. (*124*) used simulated rain and observed that several conservation tillage systems reduced sediment loss but increased the loss of soluble nitrogen in the runoff. Ketcheson and Onderdonk (*74*) found that covering broadcast fertilizer with a chopped cornstalk mulch reduced soil phosphorus losses 65 percent and fertilizer losses 97 percent.

No-till or zero tillage (*7*) is one of the most effective practices for reducing erosion. The effect of these practices on the amount of runoff is variable; sometimes runoff is increased and sometimes it is decreased. Smith et al. (*136*) reported that nitrogen in runoff was not greatly affected by the no-till practice, whereas phosphorus increased 5- to 8-fold, probably from leaching of residues. Since runoff is sometimes decreased, nitrate leaching can be increased (*158*). Schwab et al. (*131*) found little difference in the nitrogen and phosphorus content of tile drainage from conventional and no-tillage plots.

Cropping

Sod reduces runoff and permits very little erosion. Therefore, on an average annual basis, the rotations with sod should show a reduced nutrient loss. Results from corn-wheat-clover plots (*102, 134*) indicate that the rotation reduces the total N and P loss 3- to 6-fold compared to continuous corn or wheat. Schuman et al

(*129, 130*) reported that a sod pasture lost about 10-fold less nitrogen than continuous corn. The phosphorus losses from the pasture were lower by a factor of two, even though the concentration in the runoff and on the sediment was higher. A higher concentration of phosphorus in the runoff, particularly snowmelt, from pasture or hay lands has been reported by several workers (*27, 161, 176*).

When little or no residues are left on a field, as when corn is harvested for silage, then planting a cover crop such as a small grain will protect the soil during the winter. Smith et al. (*136*) recorded a 50 percent reduction in runoff and a 40-fold reduction in losses of sediment, total nitrogen, and total phosphorus when corn was planted into a ryegrass cover crop. They found little effect on the soluble nutrients lost in runoff, which were already low.

Supporting Practices

Several erosion control practices such as contouring and terraces are physical rather than agronomic. They can be used by themselves with regular cultivation or in conjunction with reduced tillage systems to achieve even greater reductions. Bedell et al. (*11*) reported that contouring a corn-wheat-meadow rotation reduced sediment, nitrogen, and phosphorus losses from 3- to 5-fold on all crops of the rotation. Schuman et al. (*129, 130*) found terraces reduced water, sediment, and total nitrogen losses a little over 10-fold and phosphorus losses a little less than 10-fold compared with contour tillage. The enrichment ratio doubled from 2 to 4 and the phosphorus concentration in solution doubled.

Farm ponds are constructed for a variety of reasons such as stock watering, recreation, etc. They are also an effective trap for sediment and nutrients (*117*). In some soils, the ponds are difficult to seal and a local seepage problem can be created.

Nutrient Management Practices

Erosion control practices will probably solve most of the phosphate pollution problems and many of the nitrogen pollution problems. These practices will have less effect on controlling nutrients dissolved in runoff than in sediment. They have no effect and may even aggravate a nitrate leaching problem. In these cases, it is necessary to use alternative or additional practices to achieve the desired degree of control. These practices involve changing the use of nutrients. Table 4 contains a list of these practices and some of the references used in the following discussion.

Eliminating Excessive Fertilization

For preventing nitrate leaching, Olson (*116*) suggested that only enough nitrogen be applied to satisfy the crop needs, that the soil's capacity for producing nitrate be accounted for, that the nitrate already present in the root zone be taken into account, and that adequate levels of other nutrients be supplied so that there is maximum efficiency. These suggestions can be reduced to a single concept of eliminating excessive use of fertilizer.

Nitrate builds up in the soil when excessive levels of nitrogen fertilizers are used (*150*). But when only adequate amounts are used at the proper time, little of the nitrogen is left after harvest (*95, 99, 114, 135*).

The greatest difficulty in preventing excessive fertilization is in predicting what levels of fertilizer should be applied so that the resulting level in the soil is adequate. The first requirement is to predict the potential yield of the crop and thus the nutrient requirements. Then, the soil's ability to meet these requirements must be evaluated. Finally, the efficiency of the applied fertilizer in meeting the remaining nutrient requirements must be considered.

This difficulty in accurately predicting fertilizer needs and low nitrogen costs has led some growers to overfertilize so that lack of nutrients would not limit yields. Recommendations based simply on "maintenance" or "balance" approaches to replace nutrients removed by the crop should be discouraged (*120*). They fail to account for either the nutrient supplied by the soil or the losses of applied fertilizers.

The yield of any crop and its response to applied fertilizer depends upon many different soil, plant, climatic, and cultural factors (*159*). For example, experiment station reports from Maryland and Michigan show the yield of corn can vary 2-fold across a single state. Both climate and soil properties can be involved. As the precipitation during the growing season decreases, the water stored in the soil when the plant starts to grow becomes the major yield determinant.

Stanford (*141, 143*) has published extensively on estimating nitrogen fertilizer requirements. He argues persuasively that there is an internal nitrogen requirement of the crop for the expected yield. To adequately estimate this requirement requires considerable field work. A first approximation can be made by considering the expected yields and nutrient contents (Table 17, Vol. I). Good farmers in fertile farming areas will probably produce higher yields but it is anticipated that the nutrient content of the crops will be proportionately

Table 4. Bibliography for nutrient management practices (Volume I, Section 4.3).

Nutrient management practice			Citations	Significant subjects
No.	Page No. in Vol. I.	Description		
General				
N1	76	Eliminating Excessive Fertilization	Herron *et al* (60)	Nitrate already in soil
			Linville and Smith (95)	Little left with adequate amounts
			Petersen and Sander (120)	Discourages maintenance and balanced approaches
			Stanford (143)	Estimating N fertilizer requirement
			Stewart (150)	Build-up with excessive use
			Thomas and Hanway (159)	Response to fertilizer
			Thomas and Peaslee (160)	Phosphorus recommendations
			Viets (168)	Implication of banning all fertilizer
Leaching Control				
N2	78	Timing Fertilizer Application	Aldrich (1)	Conditions for fall fertilization
			Bouldin, Reid, and Lathwell (22)	Argument for summer sidedressing
			Lathwell, Bouldin, and Reid (91)	Period of maximum use
N3	79	Using Crop Rotations	Bezdicek, Mulford, and Magee (12)	Soybeans don't need fertilizer N
			Olsen (113)	Profile N proportional to amount applied
			Stewart, Viets, and Hutchinson (152)	Alfalfa removes deep N
N4	79	Using Animal Wastes for Fertilizer	Ashraf (6)	Cost of storage
			Uttormark, Chapin, and Green (166)	N and P lost in snowmelt
			Zwerman *et al* (185)	Manure increases infiltration
N5	80	Plowing-under Green Legume Crops	Lyon, Buckman, and Brady (97)	References for amount of fixation
N6	83	Using Winter Cover Crops	Frink (46)	Reduced leaching by cover crops
			Thomas (156)	Recommended planting time
N7	83	Controlling Fertilizer Release or Transformation	Anonymous (4)	Cost estimate
			Boswell and Anderson (19)	Field experiment with inhibitors
			Broadbent (25)	Poor future prospects
			Hauck and Koshino (58)	Advantages of slow release

Table 4 (continued)

Nutrient management practice			Citations	Significant subjects
No.	Page No. in Vol. I.	Description		
Control of Nutrients in Runoff				
N8	83	Incorporating Surface Applications	Timmons, Burwell, and Holt (162)	Plow-down reduces fertilizer loss
N9	83	Controlling Surface Applications	Wagner and Jones (171)	Less-frequent P and K applications needed on fertile soils
N10	83	Using Legumes in Haylands and Pastures	Allison (3)	Grass uses N from legumes
Control of Nutrient Loss by Erosion				
N11	83	Timing Fertilizer Plow-down	None	

higher. More accurate data for the area under consideration are usually available from the State experiment station.

Given a yield estimate and a crop requirement, the next step is to estimate the amount of nitrogen the soil will supply without fertilizer. There are several factors to be considered. One is the capacity of the soil to produce nitrate by mineralizing organic nitrogen in the soil. An incubation method would appear to give a reliable estimate of the potential (*144, 148*) which is adjusted for temperature and moisture effects (*146, 147*). However, the time required for the incubation prohibits soil testing laboratories from using it (*33*). A hot water or steam extraction of the soil sample may provide an adequate estimate of the potential mineralizable nitrogen (*139, 142*).

Also to be considered is the amount of nitrate already in the soil (*60, 61*). In the more humid regions, any nitrate remaining in the soil after harvest will be leached out of the root zone before the crop can use it the following season. But in the more arid areas such as the Great Plains, this leaching doesn't occur regularly. A final factor that needs to be considered for the soil supply is the amount of nitrogen that is available from residues and/or cover crops.

The final step is to estimate the fraction of the fertilizer that the crop will use. Many field experiments show that the plant takes up less than 70 percent and often less than 50 percent of the applied nitrogen fertilizer (*91, 96, 116*). Some of the fertilizer is immobilized into organic matter, some is denitrified, and some can be leached out of the root zone. These changes can be reduced to some extent by applying the fertilizer when the plant is growing.

The behavior of nitrogen is quite complex and several estimates are required to predict the amount of fertilizer needed. A simpler, but often less accurate approach, is presently used in most cases. This approach relies on the results of previous experiments in the area where different rates of nitrogen were applied to the crop. Figure 5 is a summary of such an experiment (*66*). In this particular case, the soil and residues supplied enough nitrogen for a yield of 65 bu/acre. Applying nitrogen fertilizer at the rate of 120 lbs/acre produced 141 bu/acre, which was close to the maximum yield observed (147 bu/acre). Such information would be the basis for recommending that 120 pounds of nitrogen be applied to corn under similar conditions.

The phosphorus cycle is less complicated than the nitrogen cycle and the phosphorus fertilizer recommendations are also much easier to make (*160*). While less than 20 percent of the applied phosphate is usually taken up because of the reactions with the soil, there are essentially no losses by leaching or volatilization. In addition, soil tests have been extensively correlated with yield responses so that the fertilizer requirement is more readily predicted from soil tests.

Reducing fertilizer application to a less-than-adequate level doesn't always decrease pollution and may in fact increase it. Inadequate fertilization decreases growth and can increase runoff, erosion, and leaching. Smith (*134*) reports a 9-fold increase in nitrate loss from inadequately fertilized corn. Viets (*167*) discusses the implications of banning the use of all fertilizer. The effect would range from very little with soybeans in Iowa to over a 90 percent reduction in per acre yield of grapefruit in Florida. Acreage would need to be increased 20 to 30 percent for the major crops of corn, wheat, and cotton in the first year while there was still some residual fertility. In addition, the added acreage would be of lower fertility and more erodible, thereby creating additional problems.

Mayer and Hargrove (*106*) used an economic model to examine the impact of restricting fertilization to certain levels nationally and only in Iowa. Reduced use throughout the country would eliminate foreign exports, increase cropland acreage, and increase prices of farm products. If only a single state such as Iowa restricted fertilizer, the impact on the farmer would be very great, since his yield per acre would be reduced but the price for his product would not go up because of supplies from adjacent states.

Timing Fertilizer Application

The time of the fertilizer application can be an important tool for increasing the efficiency of fertilizer use and reducing fertilizer loss. Fertilizer nitrogen use is maximized when fertilizer is applied near the time of maximum vegetative growth (*21, 91*). Most crops grow the fastest several weeks after the plant emerges, as illustrated by corn in Figure 33 of Volume I (*56*). The application of nitrogen fertilizers several weeks after the plant has started to grow is commonly called summer sidedressing. Bouldin et al. (*22*) provide a number of arguments for the summer sidedressing of corn based on experiments in New York. Since the fertilizer is used more efficiently, less fertilizer is needed and the lower fertilizer cost offsets the added cost of application. They argue that if the field is too wet for sidedressing, then previously applied fertilizer will probably be lost by

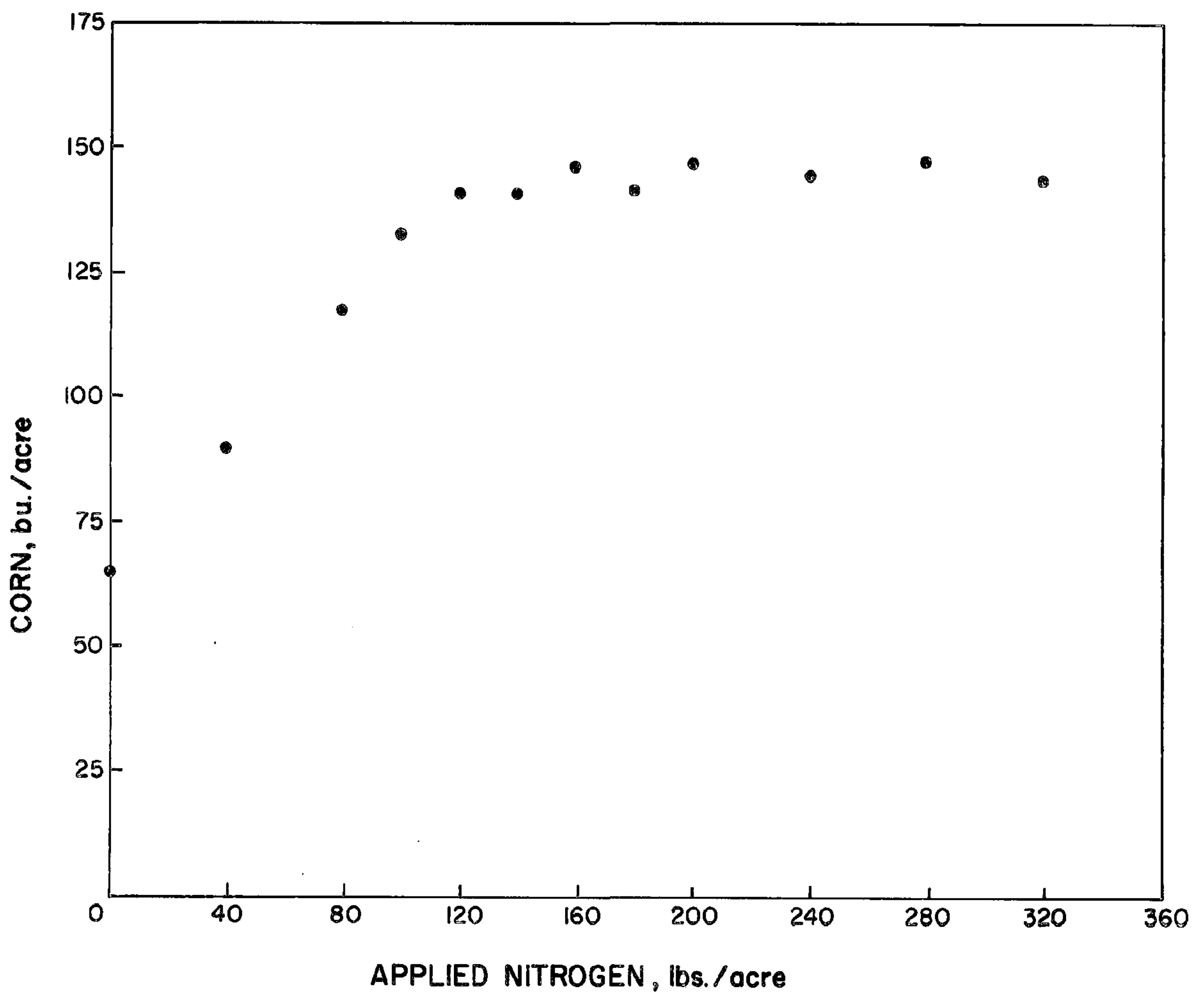

Figure 5.–An example of the yield response of corn to applied nitrogen (66).

leaching or denitrification. Conversely, if it is too dry to move the nitrate down to the roots, then the growth will be retarded by lack of water anyway.

If leaching is not a problem, then applying fertilizers preplant in the spring or even in the fall (except on sandy soils) may be acceptable. For fall fertilization it is usually recommended that an ammonium type fertilizer be applied after the soil cools below 50° F (*1*). This recommendation is based on the facts that while nitrate is mobile, ammonium is relatively immobile and is converted to nitrate very slowly below 50° F (*40, 126, 182*).

Nelson and Uhland (*112*) were among the first to show regional variation in leaching. Using Thornthwaite's calculations with a constant 4 inches of water-holding capacity and implicitly accounting for the temperature effects on nitrification, they divided the area east of the Rocky Mountains into four regions of different leaching potential (Fig. 20, Appendix B).

We have attempted to provide a more detailed mapping by combining a nitrification model with a percolation model. These models are described in detail in Appendix B. The results of the simulations are also presented in Appendix B as maps of percentage loss for various soil groups. These maps can be used to estimate average leaching losses for an area if the hydrologic characteristics of the soils are similar to one of the groups modeled. Losses of fall-applied and spring-applied ammonia (Figs. 34 and 35, Vol. I) by nitrate leaching were prepared from these maps by selecting the appropriate loss for the predominant soil group in each Land Resource Area.

These results still represent coarse approximations in spite of the numerous factors that were incorporated into the model. Only a few combinations of soil characteristics were modeled. Another limitation was the relation used for the ammonium-nitrate conversion. Other factors such as moisture and pH could modify the actual conversion rate for a particular soil. Also, immobilization and denitrification could remove some of the nitrate produced, and the leaching process may not be exactly plug flow. The model assumes little transpiration and no uptake during the winter, which would tend to overestimate leaching in southern states where there could be plant growth. Thus, these maps can serve only as first approximations and more detailed information on locally important variations must be obtained from the Soil Conservation Service, State experiment stations, and extension staffs for each area.

Split applications, applying part of the nitrogen in the fall or spring and the rest as a summer sidedressing, combines some of the features of each time of application. The first application provides enough fertilizer for a poor year. The last application is not large enough to cause toxicity problems that sometimes occur and it provides an opportunity to adjust for favorable weather, increased plant population, and optimum planting dates.

Using Crop Rotations

Crop rotations can be used to reduce the average amount of nitrogen fertilizer required. High nitrogen-requiring crops such as corn, cotton, and sorghum can be rotated with crops requiring less nitrogen, such as small grains, or legumes which require only small amounts of starter fertilizer, such as soybeans or alfalfa. Olsen (*113*) found the amount of nitrogen in the profile was proportional to the amount of nitrogen applied during the rotation. Thus, the average nitrogen content in drainage from a watershed with diversified crops should be lower than if the watershed were completely in crops like corn.

Alfalfa is particularly useful because its deep root system can remove some nitrate from deeper depths than most crops can (*152*). Soybeans are high cash value legumes that don't require nitrogen fertilization but appear to respond to high levels of soil nitrogen from previous crops (*12, 175*). The major limitations in crop rotation are the loss of cash income and/or the cost of additional equipment.

Using Animal Wastes for Fertilizer

Animal wastes, or manure, have been used as a source of plant nutrients for thousands of years. A previous section discussed many of the properties of manure. Zwerman et al. (*184, 185*) report the increased infiltration and reduced runoff from long periods of manure use. This section will be concerned with the problems associated with using manure as a substitute for fertilizer.

The most serious problem from a water quality standpoint is the loss of nutrients in runoff. Animal manure produced during the winter must be either stored or applied when the crops are not growing and chances of loss are greater. Since equipment can't enter fields that are wet with fall or spring rains, farmers with little or no storage capacity are forced to spread the manure on frozen or snow-covered fields. The resulting runoff from rains or snowmelt can carry 10 to 20 percent of the nitrogen and phosphorus in the manure (*109, 166*). The losses from a manure application containing 100 pounds of N per acre are 10 to 20 pounds of N per acre and 3 to 10 pounds of P per acre (*107*).

Plowing-down manure soon after application is the most appropriate method of controlling losses from broadcast applications. This method also prevents nitrogen loss as volatile ammonia. However, meadows and haylands can't be plowed, nor can frozen croplands. For these cases, the State of Maine guidelines (*101*) recommend that only upland fields with less than 3 percent slope be used for manure spreading when frozen or snow covered. They also recommend against spreading on any fields with slopes greater than 25 percent or within 100 feet of wells, springs, ponds, or lakes, or when there is a high possibility of runoff.

While most manure is in a relatively solid form, some stored wastes are in a slurry form. Slurries, principally from dairy and swine operations, can be injected directly into the soil, thus almost eliminating runoff losses. A large part of the nutrients are in solution and can move into the soil even if surface applied. Storage of manure is a large added expense with little economic return. The investment needed for manure storage on a dairy farm has been estimated as three to five times the cost of daily spreading (*6*).

A second problem of substituting manure for fertilizer is determining how much nitrogen is being applied. Not only does the nutrient content of manure change with different animals, but also with their feed and environmental conditions. If bedding is used, this provides added bulk with no added nutrients, but it absorbs liquids and prevents nutrient loss. The largest losses are probably by volatilization of nitrogen during storage, spreading, and before incorporation. As much as 40 percent of the nutrient value can be lost by delaying incorporation for 4 days (*127*). Ammonia gas is continually being lost, while denitrification of nitrate and nitrite occurs in anaerobic storage.

Other problems include the fact that not all the nitrogen is available during the first growing season. Organic nitrogen compounds are similar to the soil organic matter in that some are more easily mineralized by microbes than others. This is not too serious since repeated yearly applications will build the organic nitrogen up so that the total mineralized is equivalent to the amount added. The nutrients may not be in the proper ratio for a particular crop on that field. However, this can easily be corrected by adding nutrients to the field or to the manure in storage.

Plowing-Under Green Legume Crops

This practice was frequently used to supply nitrogen before the development of commercial fertilizers, but little research has been done on it since commercial fertilizers became available. Thus, the principal sources of information are older soil science books (*97, 108*). The practice is based on the symbiotic relation between some types of microorganisms and legume plants in which nitrogen from the atmosphere is converted into plant protein. From 40 to 60 pounds of N can be supplied per ton of dry forage. Part of the nitrogen in the plant comes from mineralized soil nitrogen, but this is probably balanced by not considering the nitrogen in the plant roots. Obviously, the greatest limitation of this practice is loss of any return from this crop. If the forage is harvested, then the net gain in soil nitrogen is small.

Using Winter Cover Crops

Although winter cover crops are recommended for control of soil erosion during the fall and winter (see Erosion Control Practices), they can also reduce nitrate leaching through plant uptake of nitrate and reduce percolation by drying the soil out. An oat crop reduced nitrate leaching 4-fold on one soil and eliminated it on another. Vetch, however, reduced leaching only slightly and because it is a legume, added nitrogen (*14*). Cover crops of oats, timothy, and rye reduced leaching 40 to 60 percent (*46*). Thomas (*156*) recommends that the cover crop be planted by October for the most effective control of leaching.

The major expense of this practice is planting the crop. Some economic return can be obtained by using the crop for winter grazing. Also, in some areas it is possible to double crop; that is, grow a winter grain such as wheat and then plant a short season crop such as soybeans. A serious limitation of cover crops is that they can remove so much of the soil water that the main summer crop suffers, particularly in a dry year.

Controlling Fertilizer Release or Transformation

Many researchers have explored the possibility of controlling fertilizer release or availability. Recently the interest has been very great and over 50 papers were presented at the 1974 Annual Meeting of the American Society of Agronomy dealing with this subject. Two basic approaches are being used: a slow release fertilizer and a nitrification inhibitor.

Slow release fertilizers offer three advantages: i) reduction in nutrient loss by leaching and runoff, ii) reduction in immobilization before plant uptake, and iii) reduction in losses by denitrification and volatilization (*58, 77, 119, 122*). Three processes are used to slow the release of the fertilizer from the granule: i) controlling dissolution by a physical barrier, ii) using compounds of limited water solubility, and iii) using a barrier that decomposes. Of the 13 different slow release fertilizers developed, sulfur-coated urea seems to be the most promising. Most of the commercial production is being used on turf. The present cost is 25 to 40 percent more than uncoated urea (*4*). The greatest problem is to control the release so that the fertilizer is available when the plant needs it. If the release doesn't occur in a short period of time for row crops, then the remaining N is susceptible for leaching after harvest. The future looks promising, but more research is needed before large scale recommendations can be made.

Nitrification inhibitors are chemicals that prevent microbes from converting ammonium to nitrate. Five chemicals offer possibilities. The most widely tested are: 2-chloro-6-(trichloromethyl) pyridine, sold as N–SERVE by Dow Chemical; 2-amino-4-chloro-6-methyl pyridine, sold as AM by Mitsui-Toatsu Industries; and sodium azide. Experiments with soil in plastic bags in the field from November to April show that most of the conversion was prevented by N-SERVE (*19*). One of the greatest difficulties has been to keep the inhibitor near the ammonium; usually percolating water separates them. Broadbent (*25*) doesn't consider the prospects of developing a practical method to be very good.

Incorporating Surface Applications

Immediate incorporation of surface-applied fertilizers and manure can prevent significant losses of nutrients. A number of studies have shown that the losses are greatest when the runoff occurs soon after application (see the section on runoff losses). Timmons et al. (*162*) report that deep incorporation of the fertilizer by plowing down and subsequent disking reduced the nutrient losses to levels similar to those in runoff from unfertilized plots. Broadcasting on a plowed surface is adequate if no additional tillage is performed because the infiltration is very high. Disking instead of plowing broadcasted fertilizer was not effective. Up to 30 tons/acre of manure have been incorporated into the soil with little increase in the nitrate and ammonium contents of the tail water from irrigation (*151*).

Controlling Surface Applications

The time of application and the type of fertilizer can be controlled to some extent. Fertilizer should not be applied during periods of expected runoff. Fall-seeded grains are often top-dressed with fertilizer in the spring. If leaching is not a problem, then fertilization at planting would reduce runoff losses. Pasture and haylands usually require surface application of fertilizers and thus runoff losses could be a problem. Wagner and Jones (*171*) report that if a high level of fertility is maintained, then timing of phosphorus and potassium fertilizer applications is not critical. In fact, on slightly deficient soils enough P and K can be plowed under when the forage is planted to last 4 or 5 years. Since nitrogen is so mobile, the greatest efficiency is obtained by applying it shortly before or during the growing season. Cool season forages such as the brome and blue grasses make their growth in the spring and fall, whereas warm season grasses such as the Bermudagrasses make their best growth during the summer.

Spraying a fluid fertilizer on the surface might have lower losses in runoff than broadcasting granular fertilizers even though there is no published research on the subject. Fluid fertilizers are liquids or suspensions of micro crystals and therefore should come in quicker contact with the soil than granules that must dissolve. Ammonia cannot be used because it volatilizes too easily. Urea is also subject to some losses because it is converted to ammonium.

Using Legumes in Haylands and Pastures

Legumes can be planted with grass in pastures and haylands to supply much of the nitrogen requirement and thus reduce the need to fertilize with nitrogen. The legumes can be very effective in this situation because the grass receives nitrogen in leakage from the legumes as well as the mineralized soil nitrogen (*3*). As pointed out by Kilmer (*76*), leaching losses will be higher from legumes than from grasses. However, if leaching is not a problem and runoff losses are, then legumes can be used effectively. A major problem is that competition for other nutrients, water, and sunlight causes the grass to crowd out the legume in a few years.

Timing Fertilizer Plow-Down

When nutrients are being lost by sediment transport, erosion control practices are the obvious answer in most cases. An additional procedure that can be recommended is plowing during the least erosive period and leaving the field in the least erosive condition. For example, if erosion is less in the fall than in the spring, phosphorus and potassium fertilizer might be plowed under in the fall using stubble mulching techniques or followed by a cover crop so that an erosive period in the spring can be avoided.

Mathematical Models

The soil-plant system is very complex and dynamic and therefore the impact of various management practices can vary considerably. One of the most efficient ways of testing alternatives on complex and dynamic systems is to describe the system with a mathematical model. If the model adequately represents the behavior of the system for known responses and is based on sound fundamental relations, then its responses for various alternatives can be quickly tested with modern computer equipment.

A number of programs have been started in recent years to develop models for sediment and chemical transport from watersheds. Most of these efforts are still in the development stage and have yet to be adequately tested against field results.

ACTMO, an agricultural chemical transport model (*44, 45*) linked together a hydrology model (*65*), an erosion model (*118*), and a chemical model (*42*) to predict the concentration and amount of pesticides and nutrients on a storm-by-storm basis for a farm-sized watershed. The hydrology model has been tested in a number of locations. The erosion model is being tested and improved with Corn Belt watersheds, but the chemical model is essentially untested.

Hagin and Amberger (*54*) developed a model for the N and P loads in water and have tested only a few of the various relations used. Kling (*81*) developed a model for sediment and phosphorus movement that was tested at locations in New York and Pennsylvania. Johnson and Straub (*72*) developed more of an accounting system than a model for a 23-square-mile watershed in Minnesota. EPA is developing a pesticide transport model (*31*) and a nutrient transport model.

As these and other models (*43*) are developed further, they will provide increased understanding of the system. When they are completed, a tool will be available to better evaluate the alternatives for controlling pollution from nonpoint sources. Until then, we are left with the less exact recommendations discussed in this report.

RESEARCH NEEDS

The most immediate need is to measure and evaluate the impact of presently used conservation practices on chemical losses. It is anticipated that in most cases the needed control can be achieved by practices already developed. The other cases will require a better understanding of the system to develop adequate control practices. Many of the conservation practices were developed for the older farming systems. They are now being modified and adapted to the current large-scale farming systems. The evaluation process for these modified practices should include quantitative measurements of chemical transport as well as the traditional variables such as erosion, crop yield, etc.

All forms of fertilizer research received less than adequate support in the previous decade. Now the combined effects of environmental concern, increased fertilizer costs, and increased food needs dictate that we improve our knowlege of the soil-plant-water system. Not only do we need to know the total amount of nutrients needed by various plants, but also when they are needed. We must be able to predict the rate of nutrient release from the soil as well as the total amount. In order to improve fertilizer efficiency we need a better understanding of the effects of placement and timing.

New fertilizer materials need to be evaluated in terms of leaching and runoff characteristics as well as crop yield and cost. Continued development is needed on chemical regulators such as nitrification inhibitors that can provide additional management control.

The demand for low cost meat should increase the need for more forage of higher quality. The use of commerical fertilizers, animal wastes, and legumes in achieving improved forage must be evaluated in terms of environmental quality as well as costs. The possibility of improving nitrogen fixation by legumes should be pursued. Finally, the possibility of incorporating nitrogen fixation into nonlegumes, while remote at this time, could be so useful that it should not be neglected.

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CHAPTER 5

PESTICIDES IN AGRICULTURAL RUNOFF

J. H. Caro

The use of chemical pesticides has resulted in enormous benefits to mankind, chiefly in the areas of public health and agricultural production. In public health, insect control programs have saved millions of lives by combatting such diseases as malaria, yellow fever, and typhus. In India, for example, the use of DDT has been credited with increasing the average life expectancy from 32 to 47 years of age. In Sri Lanka, annual malaria cases dropped from 2 million in 1950 to 17 in 1963; when use of DDT was then discontinued, the number of cases immediately rose and again reached 1 million in 1968 (*45*). Agricultural benefits are many: pesticide chemicals have promoted higher crop yields and improved quality of produce; aided the mechanization of agricultural production, with substantial reduction in labor requirements; and have helped to improve the utilization and management of land. Use of agricultural pesticides also has resulted in important economic benefits to both the farmer and the consumer of food and fiber. Insecticides are widely estimated to return $5 to the farmer for every $1 expended (*127*), which often tips the scales to economic profit from a crop rather than economic loss. Agricultural products would probably cost the consumer two or three times more than at present if the use of chemicals were eliminated (*150*).

Despite all the far-reaching benefits, the use of pesticides has brought about a conflict of interest because of the possibility of harmful impact on environmental quality. Conservationists have often indicted pesticide residues as being responsible for a variety of injurious effects, including fish kills, reproductive failures in birds, and acute illnesses in man and animals. Agricultural applications have just as often been charged with being primary sources from which the chemicals dissipate into the environment. Although acute adverse occurrences have indeed taken place, the sources of the damaging pesticides have been a matter of some dispute. With respect to chronic effects, the true significance of low residue levels of most pesticides in the general environment resulting from long-term use of the chemicals is still not well understood. In any event, it is widely expected that the use of chemical pesticides will remain an integral part of agricultural technology for many years and will in fact increase at least through the next decade. Consequently, information on the pathways by which pesticides leave the site of application and distribute throughout the environment will continue to be actively sought so that appropriate controls can be instituted.

One such pathway is the movement of pesticides away from treated fields in runoff water and on sediment carried along in the water. In Volume I, we presented guidelines for identifying areas of potential pollution problems arising from this movement and also described appropriate pesticide management practices that would alleviate the problems. In this chapter, we will provide documentation to support the recommended practices. We will also indicate the size of the potential problem by showing the extent of agricultural use of pesticides, and we will examine the state of knowledge concerning pesticide transport in runoff. Related areas to be covered include (1) information on pesticide persistence in soil, which affects the relationship between amounts of residues moved in runoff and the time elapsed since application of the pesticide to the field; (2) characteristic levels of pesticides found in the aquatic ecosystem; and (3) the impact of pesticides on aquatic organisms, which will permit some assessment by the reader of potential hazards of the reported levels. In addition, we will summarize information on methods for removing pesticide residues from the aquatic environment, and we will show the areas within the broad subject of pesticides in runoff that clearly require additional research.

EXTENT AND TRENDS IN USE OF AGRICULTURAL PESTICIDES

Because of the demands of our increasing population for space for cities, roadways, and recreational areas, and because economic trends have made smaller and less efficient farm units unprofitable, the amount of cropland that supports each of us has, until very recent times, declined steadily since the 1920's (Figure 1). The same pressures have favored ever more intensive farming of the acreage still cultivated. An important component of this modern, high-efficiency agriculture is the use of chemicals to combat the pests and blights that attack our crops and agricultural products. The insects, weeds, and plant diseases that cause significant agricultural damage are many and varied. In the United States alone, for example, there are an estimated 10,000 species of damaging insects and mites, about 600 of which are serious pests that require control every year (*114*).

The chemicals employed to control these pests have been highly effective. The general impact of pesticide use on crop yields is indicated in Table 1, which shows an apparent relationship between rates of pesticide application and crop yields in major geographic areas. Although yields may be increased by any of a number of improved agricultural practices, the importance of pesticide use is undeniable.

The number of specific chemicals is impressive. The current domestic market for pesticides includes more than 1800 biologically active compounds sold in over 32,000 different formulations. In 1971, 833 million pounds of the compounds were used in the United States, of which almost 60% were accounted for by use on farms (Table 2). Of all pesticides used on farms, herbicides comprised 52%, insecticides 39%, and chemicals for control of plant diseases 9%.

The extent and distribution of pesticide use on major crops in 1971 is shown in Table 3. Three crops–corn, soybeans, and cotton–accounted for almost 80% of all herbicide use on farms; two crops–cotton and corn–accounted for nearly 70% of insecticide use; and fruit, nut, and vegetable crops accounted for 85% of fungicide use. The most extensively treated crop was peanuts, whereas only small percentages of alfalfa acreage were treated. The pests of most concern in specific crops are shown clearly in the table. Weeds, for example, are a severe problem in soybeans, but insect damage is limited. Conversely, tobacco generally requires protection from insects but not from weeds.

The geographic distribution of cropland treated with pesticides is shown in Table 4. Although the table shows

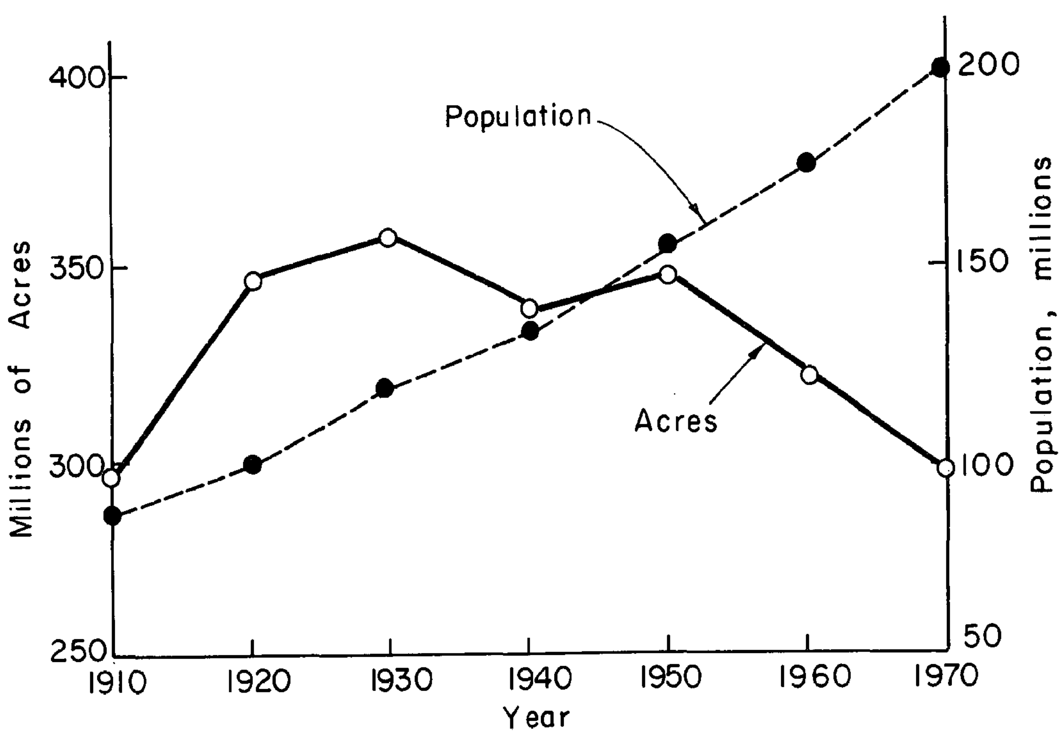

Figure 1.–Land in crops and population in the United States. From Barrons (17).

Table 1. Rates of pesticide application and yields of major crops in countries and geographic areas[1]

Country or Area	Pesticide application rates	Yields of major crops
	Ounces/A	*lb/A*
Japan	154.0	4,890
Europe	26.7	3,060
United States	21.3	2,320
Latin America	3.1	1,760
Oceania	2.8	1,400
India	2.1	730
Africa	1.8	1,080

[1] From Ennis *et al* (61).

Table 2. Use of pesticides and percentage used by farmers, United States, 1971[1]

Type of pesticide	Million pounds used[2]	Percentage used by farmers
Herbicides	359	63
Insecticides	319	53
Fungicides	155	27
Total	833	59

[1] From Andrilenas (5).
[2] Active ingredients.

data for 1969, the relative distribution has probably not changed significantly in succeeding years. By far the largest acreage treated is in the two North Central regions, which basically comprise the Corn Belt and much of the Wheat Belt. These 12 states contain 63% of the cropland receiving herbicides and 47% of that receiving insecticides. The treated area is lowest in the Northeast (New England and Middle Atlantic regions).

In the United States, the use of pesticides has continued to increase. Total annual sales for domestic use exceeded one billion pounds in 1973 (*6*), as contrasted with 833 million pounds in 1971 (Table 2), and much of the increase is undoubtedly attributable to expansion in agricultural demand. There are two main reasons for this. Within the past year or two, farm prices have risen, so that the crop is worth more to the farmer and he will tend to apply pesticides at a lower level of infestation to protect it. Second, and also as a result of changing farm economics, land is once more being converted to cropland after a long period in the reverse direction (Figure 1). The U.S. Department of Agriculture has projected an annual increase in cropland within the near future of approximately 4%. This corresponds to over 3.5 million additional acres of corn, of which about 57% will require herbicides and 33% insecticides, and more than 8 million additional acres of wheat, of which about 2.5 million acres will need herbicide treatment.

Table 3. Acres of crops grown and percentage treated with pesticides, United States, 1971[1]

Crop	Million acres grown	Percentage of total acres treated with pesticides for control of		
		Weeds	Insects	Diseases
Alfalfa	27.5	1	8	<0.5
Corn	74.0	79	35	1
Cotton	12.4	82	61	<0.5
Fruit crops	4.0	28	81	53
Peanuts	1.5	92	87	85
Rice	1.8	95	35	–
Small grains[2]	91.7	37	5	<1
Sorghum	20.8	46	39	<0.5
Soybeans	43.5	68	8	2
Sugarbeets	1.4	75	30	13
Tobacco	0.8	7	77	7
Vegetable crops	4.8	43	62	27

[1] Adapted from Andrilenas (5).
[2] Includes wheat, oats, barley, rye, and mixed grains.

Table 4. Cropland acreage treated with pesticides, by geographic region, 1969[1]

Region[2]	Cropland acreage treated for control of	
	Weeds	Insects[3]
	1000 acres	*1000 acres*
New England	260	245
Middle Atlantic	1645	675
East North Central	20060	7890
West North Central	33375	10885
South Atlantic	4030	4210
East South Central	4265	2670
West South Central	9990	7545
Mountain	5655	1985
Pacific	5635	3780
United States	84915	39880

[1] From U.S. Bureau of the Census (154).
[2] States in the regions are:
NE: ME, NH, VT, MA, RI, CT
MA: NY, NJ, PA
ENC: OH, IN, IL, MI, WI
WNC: MN, IA, MO, ND, SD, NB, KS
SA: DE, MD, VA, WV, NC, SC, GA, FL
ESC: KY, TN, AL, MS
WSC: AR, LA, OK, TX
MT: MT, ID, WY, CO, NM, AZ, UT, NV
PA: WA, OR, CA, AK, HI
[3] Not including land for hay crops.

Some economists estimate that the use of chemical pesticides will increase up to 15% annually over the next few years (*45*) because of the economic situation and potential fuel shortages. To save fuel, farmers may well replace band treatment with broadcast applications, with a resultant increase in pesticide applied per acre. Within the growing market, the patterns of use of specific chemicals will differ from the present because of shifts in cropping patterns and pest populations and the buildup of resistance in target species of pests. In the North Central States, for example, there has been a marked movement away from production of extensive crops (wheat, oats, barley, rye, flax) and toward row crops (corn, soybeans). Between 1961 and 1973, row crops increased from 58 to 80 million acres, whereas extensive crops fell from 49 million to 34 million (*141*). Weed populations will change because the varieties that are more easily controlled will be selectively removed. Despite the difficulties in moving chemicals through the registration process, new pesticides will continue to be introduced into the marketplace to combat stubborn weeds and to counteract pest resistance to older chemicals.

DISSIPATION OF PESTICIDES FROM TREATED LANDS

During application to soil or foliage, pesticides may be lost in spray drift or by volatilization; after application, they disappear from the site of application by various pathways. The chemicals may undergo biological or chemical degradation; on foliage or the soil surface, they may degrade under the action of sunlight or they may evaporate; they may be taken up into the plant and removed in the harvested crop; they may be adsorbed onto soil particles and moved off the treated area in eroded material; or they may dissolve in rainwater (or irrigation water) and move away in surface runoff or down through the soil in the soil solution, perhaps later to reappear in surface runoff or groundwater. The rates of disappearance and the fractions moving by each pathway depend primarily on the properties and formulation of the pesticide; the type, microbial population, moisture level, and type of management of the soil; the extent and intensity of rainfall; and the soil and air temperatures.

The movement in runoff water, eroded sediment, and subsurface water is of direct concern here and will be examined in detail. A number of excellent reviews are available that deal with the other pathways: Kaufman (*98*) on microbial degradation of pesticides, Crosby (*54*) and Armstrong and Konrad (*9*) on nonbiological degradation, Spencer (*145*) and Guenzi and Beard (*76*) on pesticide volatilization, Caro (*33*) and Nash (*120*) on uptake of insecticides by plants, and Foy et al (*71*) on uptake of herbicides by plants.

Factors Influencing Pathway of Movement into Water Courses

Adsorption and Solubility

The pathway that a pesticide takes in its movement away from the site of application through the action of water—that is, whether it moves in runoff water or eroding sediment or leaches down into the soil—is governed by sorption and solution equilibria that depend primarily on the water solubility of the chemical, the degree and strength of its adsorption on soil, and on the interaction of both soil and pesticide with water (*144*). Generally, compounds that are more water-soluble will move primarily in runoff water and those more strongly adsorbed will move mostly on sediment. An inverse relationship exists between solubility and extent of adsorption, but only within families of compounds. Some pesticides, such as paraquat and diquat, are very water-soluble but will move only on the sediment because of strong, irreversible adsorption; others have low water solubility but will nevertheless move in the water except when applied to a rather adsorptive soil (*153*).

Soil characteristics are clearly very important in determining the degree of adsorption, as illustrated by the fact that adsorptivity of a pesticide can vary by as much as 15-fold over a range of soil types. The most important soil property that influences the way a pesticide partitions between soil and water, as determined by a number of investigators, is the organic matter content, which usually gives a good direct correlation with the degree of adsorption. Other properties that may be important are the acidity, cation exchange capacity, moisture content, temperature, and clay mineral content. With some pesticides such as the triazine and triazole herbicides, adsorption depends primarily on soil acidity: in acid soils, they associate with free hydrogen ions to form cations that adsorb strongly to the negatively charged soil; in neutral or alkaline soils, they are in molecular form and are held much more weakly by the soil (*163*). The acid herbicides, such as 2,4-D and picloram, also adsorb more strongly in acid soils. Bailey and White (*13*) showed correlation coefficients for a number of soil properties and adsorption as part of a comprehensive review of the adsorption and desorption of pesticides in soil.

Water in the soil competes with pesticides for adsorption sites on soil particles, so that as the moisture level in the soil decreases, the fraction of the chemical adsorbed increases (*82*, p. 100). Rain falling on a dry soil will therefore desorb a portion of the pesticide, which would then move with the water in any ensuing runoff. Generally, adsorption decreases as soil temperature increases, and the response to changes in temperature becomes less as the adsorption bonds weaken (*82*, p. 97).

Pesticides adsorb preferentially on smaller soil particles because of the high surface area per unit weight of these particles. When runoff occurs, the small particles are transported greater distances than coarser material. Since rill and sheet erosion primarily involve surface soil, such erosion will tend to favor movement of the more strongly adsorbed pesticides (*107*, p. 431). Higher pesticide concentrations in eroded material do not necessarily mean that gross losses will be greater in the sediment than in runoff water; the reverse sometimes is true because the amounts of water moved are so much greater (*134, 166*).

Leachability

Pesticides in the soil or on its surface may move down through the soil profile dissolved in water. The principal factors affecting the movement are the same as those controlling overland movement--adsorption and solubility–because a pesticide is partitioned between soil and water in leaching as well as in runoff. Other parameters influencing leaching are water flow rate and amount, and the formulation, concentration, and rate of degradation of the pesticide (*89*). The correlation of solubility and adsorption with pesticide movement through soil suggests that solubility may be important in the initial movement from the point of application, whereas adsorption may be the determining process in later movement. Therefore, adsorption will, in general, be a better indicator of overall potential movement than will solubility (*21*). As examples of this, prometryne (48 ppm water solubility) moves less in soil than simazine (5 ppm) because it is more strongly adsorbed, and monuron (230 ppm) moves about the same as atrazine (33 ppm) for the same reason (*85*).

The pesticide moves downward through the profile either by mass flow of water from impacting rainfall or by molecular diffusion in the soil solution. Diffusion, which is influenced by bulk density and temperature in addition to soil moisture, is slow in comparison with mass flow and is important only over short distances (*21*), so that mass flow is the primary means of movement under most conditions. Water does not, however, continuously move downward, except in very high rainfall areas. Evaporation at the surface causes upward movement of subsurface water and its complement of dissolved pesticides, which then concentrate at the surface (*107*, p. 429). Pesticides in the water can also move laterally when they encounter a zone of water saturation or when they reach the boundary between two areas of different soil moisture, since water will move laterally into the drier soil (*89*). If the laterally moving water intercepts the sloping surface of the land, the dissolved pesticide will join the overland flow and appear in surface runoff.

Despite occasional reports of low-level groundwater contamination by pesticides, measurements have not in general shown that groundwater pollution by leaching of pesticides through soil is extensive or significant. Much excess water must be applied even to a relatively mobile chemical to move it deeply into the profile. Many reported findings in experiments with picloram, a relatively leachable herbicide, are in agreement: except in sandy soils, picloram does not leach below the 2-foot depth (*107*, p. 430). For a more strongly adsorbed pesticide such as dieldrin, several hundred years would be required for the chemical to be transported in solution at a residual concentration of 20 ppb to a depth of 1 foot in neutral soils (*67*). The groundwater contamination that does occur may be caused by pesticides being carried on soil particles washed down into deep cracks in the soil in drought-breaking rainfalls (*125, 169*).

Formulation

The formulation in which a pesticide is applied also may affect the pathway of movement, especially if runoff occurs shortly after application, before the chemical has equilibrated with the soil. For example, ester formulations of the herbicide 2,4-D applied to a sandy loam soil in a set of experiments (*16*) were far more susceptible to washoff than an amine salt formulation. The amine formed a true solution with water and leached into the soil, whereas the relatively insoluble esters were adsorbed and moved on the eroded sediment.

Factors Influencing Amounts of Pesticides Moved into Water Courses

The quantity of a pesticide moving into a water course from a treated area in any given runoff occurrence depends on a number of associated factors. Its relationship to topography, intensity and duration of rainfall, soil erodibility, and land management and cropping practices are discussed in the chapters on erosion and runoff. Obviously, the amount moved will increase with the amount of pesticide initially applied to the area. It also depends on the following parameters.

Time After Application

Characteristically, pesticide losses are highest in the first runoff occurring after application of the chemical, and the magnitude of the loss generally decreases as time between application and runoff increases. The effect of elapsed time is particularly noticeable with short-lived pesticides and with pesticides that are not incorporated into the soil. Concentrations of the chemical in subsequent runoff events decrease at a rate that depends largely on the persistence of the pesticide in the soil. Field experiments with the carbamate insecticides carbaryl and carbofuran showed that pesticide concentrations in both runoff water and sediments in the third runoff, which occurred within 1 or 2 months after application, were less than 5% of the concentrations in the first runoff (*35, 36*). By contrast, concentrations of the persistent insecticide dieldrin in the third runoff, 3 or 4 months after application, were about 15% (water) and over 30% (sediment) of those in the first runoff (*34*). The pattern of relatively high concentrations in the first runoff, decreasing with time to eventually negligible concentrations in succeeding runoffs, has been noted in experiments with many pesticides. Quantitative examples are illustrated below in the section on pesticide levels in runoff.

Persistence in Soil

As mentioned, the persistence of a pesticide in soil affects the change with time in amounts lost in runoff. However, many factors influence the persistence of an individual pesticide and, consequently, it can be quite variable. Picloram, for example, has been reported in specific instances to effectively disappear from the soil in as little as 50 days (*110*) or as long as 6 years (*30*), but its persistence under moderate conditions is generally about 1.5 years.

A pesticide applied to the soil is subject to a sequence of overlapping loss processes—application losses, volatilization, sorption, leaching, and eventually chemical and biological degradation (*92*). As a result, the loss rate changes rapidly during the early period when the chemical is distributing and equilibrating in the soil, then becomes nearly constant over a relatively longer time. This later rate is dictated by many conditions, including among others the weather; cultural practices; and type, temperature, moisture level, and acidity of the soil. Pesticides that are subject to microbial degradation will have reduced persistence when applied to an area that had received an earlier application of the same chemical because of growth in populations of active microorganisms after the first treatment (*99*). If, as is often the case, more than one pesticide is applied, interactions between the chemicals may also markedly alter the persistence of the individual compounds in both soil and aquatic environments (*97*).

Antecedent Soil Moisture

Some pesticides will have greater losses in runoff if applied to wet soil than if applied to dry soil, particularly if runoff occurs soon after application. Experiments showing this have been reported for 2,4-D (*16*) and for fluometuron (*15*). The effect is probably related to the competition of water with the pesticide for adsorption sites on the soil particles.

Proximity to Water Course

Sloping cropland rarely abuts continuous streams. Consequently, pesticide-containing runoff usually must traverse some untreated land before reaching the water. This intervening area can trap some of the pesticide, resulting in lowered contamination of the stream. Large decreases can be obtained. In one set of measurements, flow over only 5 feet of untreated soil with slopes of 3% or 8% reduced picloram losses in runoff from small field plots more than 50% (*151*); in another case, dieldrin applied at 10 to 20 times normal levels to strips of land 12 to 15 feet away from the edges of ponds, with shallow to steep slopes intervening, did not appear in the pond water or bottom mud, except for very low (0.3 ppm) contamination of the mud when the pesticide was left on the surface of the soil (*58*).

Placement of the Pesticide

The most important effect of pesticide placement with respect to environmental contamination is that soil-incorporated pesticides will not be lost in runoff to as great an extent as those applied and left on the surface or sprayed on foliage. The subject is discussed in more detail in section 4.4 of Volume I.

PERSISTENCE AND FATE OF PESTICIDE RESIDUES IN THE AQUATIC ENVIRONMENT

Distribution of Pesticides on Entering Water Bodies

A pesticide carried in agricultural runoff entering a receiving water body—stream, pond, or lake—will distribute within the aquatic system in a manner and at a rate that depends primarily on whether the chemical is initially dissolved in the water or adsorbed on particles of eroded soil suspended in the water.

A dissolved pesticide will be diluted in the larger volume of water and will be subject to processes that dissipate it. In a flowing stream, it will simply be transported away from the point of entry, later to undergo degradation or removal from the water. In a pond or lake, it may sorb or concentrate in algae and aquatic vegetation or it may attach to suspended sediment and other particulates in the water such as bacterial flocs, diatoms, and general organic or inorganic fragmentary material. In either case, it is eventually deposited on the bottom of the lake unless it is chemically or biologically degraded before it reaches the bottom or is taken up by living organisms. The sorption processes appear to be generally quite rapid and efficient, as shown in measurements of organochlorine insecticide sorption on algae (*91*) and bacterial flocs (*106*). Highly soluble pesticides that are only weakly adsorbed may be hydrolyzed or biologically degraded in solution at a rate that depends on the types and numbers of microorganisms in the water.

The fate of pesticides entering water bodies adsorbed on sediment has been discussed in detail by Pionke and Chesters (*126*). The pesticide will distribute first with the carrying sediment, then will equilibrate with the remainder of the aquatic system. Sediments entering water bodies will segregate on a particle-size basis: in a stream, the fractionation will depend on stream velocity; in a lake, the particles will settle on the bed in decreasing order of particle size. The finer particles containing the highest concentrations of pesticides will be transported farthest and will be localized in a stream; in a lake, these particles will settle last and remain at the water-sediment interface. In large, thermally stratified lakes, density currents may control the movement and mixing of incoming sediments and settling may be very slow.

Conditions in the water body may affect the adsorption and desorption of the pesticide on the sediment. If the pH of the lake or stream is higher than that of the inflow, desorption of acidic compounds (such as 2,4-D, 2,4,5-T, or picloram) or weakly basic compounds (such as the triazine or urea herbicides) will be favored, and the reverse will be true if the pH of the body is lower than that of the inflow. Salinity in the lake will favor adsorption of acidic pesticides and desorption of basic pesticides, but the effect is generally minor. A lower temperature in the water body will increase pesticide adsorptivity, but this too is a minor effect under field conditions (*126*). If there is any oil pollution in the water, adsorbed oil will significantly concentrate the pesticides on the sediment (*87*).

Post-Distribution Processes

Pesticides never reach true equilibrium in water bodies because the systems are dynamic, with many processes continually operating to remove the chemicals from the system at rates that change as conditions change. Pesticides sorbed on bottom muds may be churned up and carried along with sediment during periods of turbulent flow or they may remain where originally deposited. Since sorption on particulate matter is generally reversible, the bottom muds provide a continuous supply of desorbed pesticides to the overlying water. In the water, the pesticides may be chemically or biologically degraded, they may reach the surface and volatilize, or they may be decomposed near the surface by the action of sunlight. (Sunlight energy is, however, probably too weak to induce much photodegradation in natural waters.) At the surface of the bottom muds, pesticide degradation is extensive. Because organic matter accumulates there, it is an area of high microbial activity. The microbial populations may consume so much oxygen that the environment becomes anaerobic, a condition that favors the degradation of many pesticides. For example, most chlorinated hydrocarbon insecticides, although normally highly persistent, will degrade at an appreciable rate under anaerobic conditions when the temperature is 20° C or higher (*91*).

Pesticide Persistence in Aquatic Environments

Measurements have been made of the persistence in aquatic systems of a large number of specific pesticides, as summarized by Pionke and Chesters (*126*), Paris and Lewis (*123*), and Eichelberger and Lichtenberg (*60*), among others. Information of a more general nature is presented here.

Organophosphorus Insecticides

As a class, organophosphorus insecticides are among the less hazardous pesticides in the aquatic system because they hydrolyze rapidly. Within the pH range 6.0-8.5, which covers most systems, most organophosphorus compounds hydrolyze within 8 to 12 days, with hydrolysis occurring both in water and on sediments (*118, 126*). Parathion is an exception, being somewhat resistant to chemical hydrolysis. It is, however, readily susceptible to microbial degradation in both aerobic and anaerobic environments. In the absence of an active microbial population, parathion remains in the aquatic environment for several months; in the presence of an active population, it is degraded in a matter of weeks (*75*).

Organochlorine Insecticides

As noted earlier, these compounds will degrade very slowly, if at all, in aerobic aquatic systems, but will decompose more rapidly in anaerobic environments through the action of microorganisms. The degradation, which generally involves a simple dechlorination of the molecule, may require up to several months for completion in natural systems. There is, however, some variation among the individual members of the class, persistence increasing in the following order: lindane, heptachlor, endrin, DDT, DDD, aldrin, heptachlor epoxide, dieldrin (*90, 91*).

Other Pesticides

The carbamate insecticides have been shown to degrade in slightly alkaline river water in less than 4 weeks (*60*), but since the stability of these compounds is pH-dependent, appreciably longer persistence would be expected in a more acid environment.

The decomposition of many herbicides is primarily biological. In aerobic lake water, for example, 2,4-D persisted for up to 120 days, whereas in lake muds, it was substantially decomposed in 24 hours once the microbial populations had adapted to the chemical (*2*). The herbicide dicamba also dissipates from water most rapidly under nonsterile conditions. The rate of disappearance depends greatly on the temperature, especially in the presence of sediments containing active microbial populations (*139*).

CHARACTERISTIC LEVELS OF PESTICIDES IN THE AQUATIC ECOSYSTEM

When pesticide-containing runoff occurs from agricultural land, the chemicals are quickly diluted in the water bodies receiving the runoff and are also partitioned among the various components of the environment—water, bottom sediments, and living organisms—so that each component eventually bears a concentration of the pesticide. The magnitudes involved have been measured by many investigators under a variety of conditions and are summarized here, including levels in the runoff itself and in drainage streams, farm ponds, lakes, and oceans, so that some appreciation may be gained of the impact of agricultural activities on water quality.

The importance of maintaining a constant surveillance of the aquatic system in the United States has been recognized by the Federal government. Comprehensive programs for continuous monitoring of pesticides in fish, estuarine shellfish, water, and bottom sediments, among other components of the general environment, are conducted by various agencies to establish baseline levels and to signal significant trends. The overall program is coordinated by an interagency committee (*122*). With respect to the quality of our waters, standards have been set for acceptable limits of certain pesticides in drinking water (Table 5). An obvious goal of pesticide control programs is to assure that the natural waters of the country are sufficiently pesticide-free to drink without purification.

Pesticide Levels in Runoff

Concentrations of pesticides leaving treated fields in runoff water and entrained sediments during the crop season following application are almost always measurable, so that there is little doubt that agriculture does contribute to the pesticide residues found in the general aquatic ecosystem. Both concentrations and gross amounts lost depend on numerous factors, including among others the intensity of rainfall, the time after pesticide application that runoff occurs, and the mode of application. Concentrations of a given pesticide may therefore differ substantially in runoff occurrences at separate locations under different sets of conditions. Nevertheless, it is useful to examine the results of specific measurements reported in the literature, not only because an understanding may be gained of the orders of magnitude involved, but also because some

Table 5. Recommended limits for pesticides in drinking water[1]

Pesticide	Recommended Limit
Organochlorine Insecticides	
	ppb
Aldrin	1
Chlordane	3
DDT	50
Dieldrin	1
Endrin	0.5(0.2)[2]
Heptachlor	0.1
Heptachlor epoxide	0.1
Lindane	5(4)[2]
Methoxychlor	1000(100)[2]
Toxaphene	5
Phenoxy Herbicides	
2, 4-D	20(100)[2]
Silvex	30(10)[2]
2, 4, 5-T	2

[1] From Environmental Protection Agency (62).
[2] Numbers in parentheses indicate limits proposed by the Environmental Protection Agency to take effect in December 1976.

informative general conclusions may be drawn. Table 6 lists a number of such measurements made on watersheds under normal agricultural conditions.

The figures in the table show that pesticide concentrations in runoff are generally, but not always, highest in the first runoff occurrence following application of the chemical. Presumably, small flows can occasionally produce higher concentrations than more intensive flows preceding them. Concentrations are always lower in runoffs occurring later in the season, irrespective of which pesticide is applied, and the reduction is generally greater for water-borne pesticides than for those carried on sediment. The table also shows that the organochlorine insecticides, as is well known, adsorb to a great extent onto sediments, leaving only very low concentrations in water. Control of erosion will therefore reduce the movement of these chemicals substantially. Atrazine, on the other hand, moves with both water and sediment. In general, the concentrations of pesticides in runoff are considerably above drinking water standards (Table 5) and must be diluted substantially in drainage streams to avoid acute harmful effects on aquatic organisms in the streams. Finally, the table shows, particularly in the cases of carbaryl and picloram, how widely runoff concentration of a pesticide may vary under different conditions.

Though amounts of a pesticide lost in runoff may vary among specific treatments, it is almost always true that the gross loss over the course of the year following application will represent only a small percentage of the amount of chemical that had been applied to the cropland. Results of watershed and field-plot experiments reported in the literature (Table 7) show the wide applicability of the relationship. Except for one experiment with atrazine in which a heavy rain occurred one week after application, total amounts lost in runoff water and sediment were always 5 percent or less of the application. This appears to hold true irrespective of the soil type or degree of incorporation of the pesticide into the soil, and applies even on relatively steep slopes. Simulated rainfall experiments, in which the amount and intensity of water falling onto a small sloping plot can be controlled, show that rainfalls of the size and intensity that might be expected to occur every year produce only small percentage losses of pesticides in runoff, but that heavier "10-year" rains may cause larger losses (*11, 166*).

Pesticide Levels in Drainage Streams

Almost all the measurements that have been made of pesticide concentrations in flowing streams draining treated areas indicate that, as might be expected, concentrations are substantially lower than in direct runoff. With many herbicides, residues were always below detectable limits in waters a few hundred yards below sprayed areas (*66, 78*). Even where detected, levels of such herbicides as 2,4-D, 2,4,5-T, dicamba, and picloram were well below median tolerance limits for trout. In one instance, the herbicide fenuron was applied at a high rate of 23 lb/A along stream channels. Only 2.4% of the application was lost in the stream water over a 27-month period, with a maximum concentration of 430 ppb following a heavy rain (*56*). Results for insecticides were much the same as those for herbicides. Phosphorus insecticides were just at detection levels in drainage streams in California (*14*). Chlorinated insecticides were generally at low but measurable levels. In streams draining sugarcane fields, waters contained a maximum of 820 ppt (parts per trillion) of endrin over a 4-year period, generally decreasing to 30 to 40 ppt 3 months after treatment. Streambed material averaged 100 ppb, with levels decreasing as the season progressed. Dieldrin, BHC, and DDT were also found in the waters in the parts-per-trillion range (*104*). In drainage streams from a commercial orchard that had received substantial applications of organochlorine insecticides, no residues were found in the water, but detectable levels of DDT compounds, dieldrin, and endrin appeared in the silt, organic debris, and bottom organisms (*117*).

Table 6. Characteristic concentrations of pesticides in runoff: maxima and rates of decrease

Pesticide	Application rate	Runoff occurrence giving maximum concentration		Maximum concentration		Runoff occurrence giving reduced concentration		Reduced concentration		Citation
		Number after application	Days after application	In water	In sediment	Number after application	Days after application	In water	In sediment	
	lb/A			*ppb*	*ppb*			*ppb*	*ppb*	
Chlorinated Insecticides										
DDT	1.5			70.0	30,000			1.0	10,000	Haan (77)
DDT	0.65	1st	1	83.0[1]		5th	46	7.0[1]		Epstein and Grant (63)
Dieldrin	1.5			70.0	30,000			1.0	30,000	Haan (77)
Dieldrin	5.0	1st	13	20.0		3rd	18	6.7		Caro *et al* (34)
Dieldrin	5.0	1st	13		14,200	10th	82		5,000	Caro *et al* (34)
Endosulfan	0.31	2nd	4	19.0[1]		3rd	14	2.0[1]		Epstein and Grant (63)
Endrin	0.3	1st	2	2.73		3rd	27	0.53		Willis and Hamilton (169)
Endrin	0.3	1st	6	5.02		3rd	23	2.88		Willis and Hamilton (169)
Endrin	0.25	1st	1	49.0[1]				8.0[1]		Epstein and Grant (63)
Methoxychlor	20.0	2nd	18	8.8		4th	33	1.0		Edwards and Glass (59)
Other Insecticides										
Carbaryl	4.5	1st	17	248	12,200	3rd	29	8.4	80.0	Caro, Freeman, and Turner (36)
Carbaryl	1.5			1220						Fahey (68)
Phorate	0.67			19.0						Fahey (68)
Herbicides										
Atrazine	8.0	1st	24	4600	6,200	2nd	35	980	3,300	Hall, Pawlus, and Higgins (81)
Atrazine	1.0	1st	24	700	950	2nd	35	180	550	Hall, Pawlus, and Higgins (81)
Fluometuron	4.0	1st		870						Wiese (167)
Picloram	3.0	3rd	86	19.0		4th	197	9.0		Bovey *et al* (26)
Picloram	2.0	1st	30	14.4		3rd	98	<1.0		Baur, Bovey, and Merkle (19)
Picloram	1.0	3rd	6	89.7		8th	30	1.0		Baur, Bovey, and Merkle (19)
Picloram	0.25	1st	10	17.0		2nd	20	<1.0		Baur, Bovey, and Merkle (19)
2, 4, 5-T	3.0	3rd	86	287		4th	197	6.0		Bovey *et al* (26)
2, 4, 5-T	1.2	3rd	22	380		5th	38	50		Edwards and Glass (59)

[1] Water-sediment mixture.

Table 7. Percentages of applied pesticides lost in runoff[1] in field experiments

Pesticide	Incorporated depth	Soil texture	Slope	Pesticide in runoff	Citation
	In.		*%*	*% of appln.*	
Atrazine	0	Silty clay loam	14	4.8-5.0	Hall (80)
Atrazine	0	Silty clay loam	14	2.6	Hall, Pawlus, and Higgins (81)
Atrazine	0	Silt loam	10-15	2.5-15.9	Ritter *et al* (134)
Carbaryl	2	Silt loam	10	0.1	Caro, Freeman, and Turner (36)
Carbofuran	3	Silt loam	9	0.9	Caro *et al* (35)
Carbofuran	2	Silt loam	10	1.9	Caro *et al* (35)
DDT	0	Loamy sand	2-4	1.0-2.8	Bradley, Sheets, and Jackson (27)
DDT	0	Gravelly loam	8	0.7	Epstein and Grant (63)
Dieldrin	3	Silt loam	14	2.3	Caro *et al* (34)
Dieldrin	3	Silt loam	10	0.02	Caro *et al* (34)
Endosulfan	0	Gravelly loam	8	0.25-0.35	Epstein and Grant (63)
Endrin	0	Gravelly loam	8	0.01-1.0	Epstein and Grant (63)
Endrin	0	Silty clay loam	0.2	0.1	Willis and Hamilton (169)
Fluometuron	0	Various	0.1-4	<3.0	Wiese (167)
Methyl parathion	0	Loamy sand	4	0.01-0.02	Sheets, Bradley, and Jackson (142)
Methyl parathion	0	Sandy loam	2	0.13-0.25	Sheets, Bradley, and Jackson (142)
Propachlor	0	Silt loam	10-15	3.1	Ritter *et al* (134)
Toxaphene	0	Loamy sand	2-4	0.4-0.6	Bradley, Sheets, and Jackson (27)
Trifluralin	6	Loamy sand	4	0.3-0.5	Sheets, Bradley, and Jackson (142)
Trifluralin	6	Sandy loam	2	0.5-0.8	Sheets, Bradley, and Jackson (142)

[1] Both water and sediment.

Pesticide Levels in Farm Ponds

Concentrations of pesticides in the waters of farm ponds adjacent to treated areas are clearly sensitive to the amount of pesticide applied in the area and the length of time between application and the first heavy rain. In a pond near cotton plots, for example, DDT and toxaphene concentrations in the water were always significant after application and were especially high when intense rain closely followed the application. Concentrations ranged from 0.4 to 13.4 ppb for DDT and from 2.9 to 65.2 ppb for toxaphene (*142*). Similarly, in measurements of the herbicide picloram in ponds at several locations in Texas, concentrations in the water ranged from 55 to 184 ppb if the first rainfall occurred within 2 weeks after application, but were only 2 to 29 ppb if the first rainfall was delayed for 6 weeks. Concentrations in all cases dropped to 1 ppb or less within 6 months (*78*).

The few measurements that have been made indicate that farm ponds are not always contaminated despite proximity to treated areas. In Virginia in 1966, when use of organochlorine insecticides was high, heptachlor was found in the water of only 10 of 35 ponds examined, at levels up to 5 ppb. In the bottom muds, the conversion product heptachlor epoxide was found in 14 of the ponds at levels of 1 to 60 ppb. As is evident, the majority of the ponds contained no detectable residues (*162*). In another set of measurements, the waters of a farm pond near a 5-lb/A treatment with carbaryl contained no detectable residues throughout the crop season, and a pond near a 0.67 lb/A treatment with phorate showed a maximum of only 4 ppb, becoming undetectable later in the season (*68*). Of course, ponds or any water body can receive relatively heavy doses of pesticides by drift or inadvertent direct spray from aerial applications.

Pesticide Levels in Rivers

Virtually no published information is available on pesticide residues in river waters of the United States showing concentrations occurring later than about 1970. Nevertheless, the pattern is clear: contamination of the streams peaked about 1966 (Figure 2) and then decreased steadily, probably right up to the present, with decline in domestic use of the most significant contaminants, the organochlorine insecticides. Concentrations have always been low and are often at trace levels, which are less than about 1 to 5 ppt for most of the chemicals.

The national situation is perhaps best summarized by examining the results of a few broad-scale monitoring studies. Lichtenberg et al (*108*) combined five annual synoptic surveys of U.S. rivers for 1964 through 1968. Except for dieldrin in 1964, no pesticide appeared in more than 40% of the samples, and the frequency of

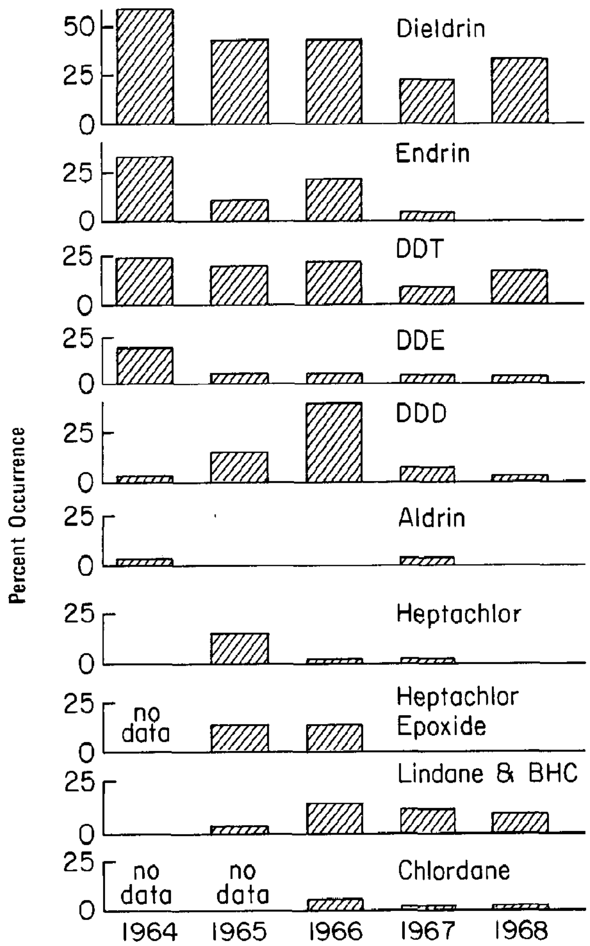

Figure 2.–Percent positive occurrences of ten organochlorine insecticides in river waters of the United States, 1964-68. From Lichtenberg *et al* (108).

positive occurrences declined sharply after 1966 (Figure 2). Maximum concentrations found during the 5-year period for the 10 compounds shown in the figure ranged from 0.84 ppb for DDD to 0.048 ppb for heptachlor. An intensive examination of Mississippi and Missouri River drinking water samples during the same period (*137*) showed frequencies of occurrence similar to those of the broader surveys: over 40% of the samples were positive for dieldrin, over 30% for endrin and total DDT, and 20% for chlordane. Little or no aldrin, heptachlor, toxaphene, or methoxychlor was found. For the later period from 1968 through 1971, measurements of organochlorine insecticides and three herbicides (2,4-D, 2,4,5-T, silvex) in waters taken from 20 stations in Western U.S. rivers (*138*) showed the characteristic decline in positive insecticide occurrences. In 1967-68, there were 165 insecticide occurrences and 70 herbicide occurrences; corresponding figures in 1970-71 were 53 and 54. Maximum concentrations found were 0.46 ppb DDT and 0.99 ppb 2,4-D. In sum, contamination of river waters by pesticides is at low levels, sporadic, and decreasing. Even at worst, residues were well below acceptable limits for drinking water (Table 5).

Pesticide residues in riverbed sediments are also relatively low. In the Mississippi River, high concentrations were found only in sediments located just below pesticide manufacturing and formulating plants (*18*). Large amounts of organochlorine insecticides applied to crops in the Mississippi River Delta did not contaminate streambed sediments widely (*112*). Dieldrin and endrin occurred in only 18 to 20% of the sediments taken from a Louisiana estuary in 1968-69; maximum levels were 4 or 5 ppb (*136*). In Rhode Island streams, measurements made about 1970 showed that DDT and its metabolites occurred in almost all sediments at concentrations generally below 500 ppb. Chlordane and dieldrin were also found, but not as often and usually at less than 50 ppb (*133*). The reported results suggest that riverbed surfaces of streams draining extensive row-crop farmland could be contaminated by locally used pesticides at low part-per-million levels.

Pesticide Levels in Lakes

Pesticide concentrations are generally lower in large lakes than in rivers. Water samples from Lake Erie in 1971-72 mostly showed no measurable residues. In the few positive samples obtained, low levels of diazinon, dieldrin, atrazine, and simazine were found. It was concluded that the contribution by agricultural sources to pesticide pollution of the lake was negligible and insignificant (*156*). In 1967-68 in Lake Poinsett, the largest natural lake in South Dakota, DDT and its metabolites predominated, averaging 80 ppt in the water, with lesser amounts of other organochlorine insecticides appearing in most of the samples (*83*). The waters in Lake Michigan contained only 2 or 3 ppt of organochlorine insecticides, chiefly the DDT family, in 1969-70. Upper sediment layers contained median concentrations of 18.5 ppb total DDT and 2.0 ppb dieldrin, with traces of heptachlor epoxide and lindane (*105*).

Being large, diverse impoundments, lakes provide excellent environments for measurement of the process of biological magnification of pesticides through food chains. Two studies clearly illustrate the effect. In one, DDT in Lake Michigan appeared at levels of about 2 ppt in the water, 14 ppb in the bottom muds, 410 ppb in sand fleas, 3 to 6 ppm in fish, and as much as 99 ppm in herring gulls, which are near the top of the chain (*114*).

In the second study, in which organochlorine insecticides in Lake Poinsett were measured, the effect was considerably smaller. Residues found were in the ratio: water 1, bottom sediments 18, crayfish 18, zooplankton 37, algae 37, and fish 790 (*83*). Whatever the magnitude, biological magnification must be considered when evaluating the significance of very low concentrations of persistent pesticides in waters and bottom sediments.

Pesticide Levels in Oceans

Pesticides in the marine environment originate from many sources and it is clearly not possible to define the contribution from agricultural runoff. Taken together, however, all sources have contaminated the ocean to only extremely low levels, and only in coastal areas. DDT, the most ubiquitous pesticide, is undetectable in Atlantic deep-sea water or sediments, but does appear in water, sediments, shellfish, and finned fish along the coast. However, concentrations in shellfish have declined to insignificant levels since the curtailment of DDT use (*88*) and, although not reported, concentrations in the other components of the ocean environment have probably also declined. Where they occur, concentrations are highest in surface slicks; in 1968, coastal slicks contained up to 13 ppb of the organochlorine insecticides, whereas the underlying waters were at low ppt levels (*140*). Whole seawater off the Pacific Coast in 1970 also contained DDT, but only at a maximum of less than 6 ppt (*50*).

The pattern of decreasing DDT concentration with distance from shore and with depth was confirmed in a more recent investigation of Pacific Ocean waters (*168*). DDT concentrations in surface film samples taken in 1971 and 1972 were 11-15 ppt in coastal waters, 0.4 ppt in the offshore California current, and less than 0.02 ppt in the North Central Pacific. In subsurface waters, by contrast, DDT levels were less than 0.01 ppt in the North Central Pacific and only 0.1 ppt in the offshore current. The measurements also showed that concentrations of polychlorinated biphenyls were ubiquitous and were as much as two orders of magnitude higher than those of DDT.

IMPACT OF PESTICIDES ON THE AQUATIC ENVIRONMENT

System planners can best evaluate the need for instituting controls for pesticides in agricultural runoff if they understand the effects that the chemicals may produce in the downstream aquatic environment. The information presented here will emphasize some major effects but does not cover all that is known on the subject.

A variety of pesticide-induced effects on aquatic birds, fish, aquatic plants, invertebrates and microorganisms has been documented. Acute responses have been measured quantitatively and subtle effects produced by long-term exposure to low, sublethal pesticide concentrations have been recognized, even though not measured directly. Perhaps the most prominent effects have been the catastrophic fish kills during the 1960's in which pesticides were implicated, though never positively established as the cause. One such took place in the Mississippi River, presumably caused by endrin; a second occurred in the Rhine River and was attributed to endosulfan. These and similar kills probably resulted from high pesticide concentrations emanating from a point source rather than from nonpoint agricultural runoff. Runoff was, however, probably partly responsible for contamination of Lake Michigan salmon by DDT in 1969 that, though not fatal to the fish, resulted in widespread confiscation of the commercial catch, with an economic loss estimated at \$3 to \$4 million (*95*).

Unless pesticide concentrations are very high, responses of organisms in a particular aquatic system are extremely difficult to predict because of the great variability and natural complexity of ecosystems and the assortment of environmental insults that man imposes on the systems. Ponds, lakes, and streams vary in their content of water, dissolved salts, temperatures, acidity, and nature and populations of plants and animals on the bottom. Pesticide contamination in small bodies of water, for example, has been judged to be more transitory and less serious than in large bodies because of greater bottom surface area per unit volume of water, higher flushing rates, and greater biological activity (*95*). Furthermore, different forms of the active pesticidal ingredient may differ in toxicity, and additives in the formulation, such as wetting agents or binders, may be more toxic to aquatic organisms than the active ingredient itself (*119*).

In general, herbicides are less toxic to aquatic organisms than insecticides (Table 8), though there are a number of exceptions (Vol. I, Tables 8a and 9a). Herbicides, especially those applied directly, also have desirable as well as undesirable effects on water bodies. They may be responsible for opening and maintenance of navigable waterways, saving of irrigation water, increase in aesthetic and monetary value of waterfront property, control of mosquitoes and snails by removal of

Table 8. Relative toxicity of selected pesticides to aquatic organisms[1]

Pesticides	Organism				
	Plankton	Shrimp	Crab	Oyster	Fish
Herbicides	1	1	1	1	1
Phosphorus . . . . . Insecticides	0.5	1000	800	1	2
DDT.	2	400	200	300	700

[1] From Butler (31). Based on the arbitrary assignment of the value of 1 to herbicides.

aquatic weeds, restoration of recreational waters, improvement in fish management, and elimination of flavors and odors from algal blooms. On the debit side, many herbicides are acutely toxic to fish (Vol. I, Table 8a). Serious losses of fish and other aquatic fauna may also occur when herbicides kill aquatic weeds, which gravitate to the bottom and decompose, removing necessary oxygen from the water. Moreover, phenols resulting from the hydrolysis of phenoxy herbicides such as 2,4-D may impart objectionable flavors and odors to water. In comparison with insecticides, however, the hazards of herbicides in the aquatic environment are small. Most herbicides have little or no toxicity to humans, wildlife, or livestock; they may reduce phytoplankton populations initially, but recovery generally occurs within 2 or 3 weeks; they do not undergo biological magnification in food chains, and shellfish are tolerant of them, accumulating residues only temporarily after exposure (*72*).

Acute Toxicity

The toxicity of a pesticide to fish is affected by numerous parameters, including the size, age, and species of the fish; water temperature and acidity; and physical differences at the aquatic site. Survival time after exposure generally correlates directly with body weight, probably because the smaller fish consume a proportionately greater diet and have less fat for storage detoxification. Higher water temperature increases toxicity of some pesticides and decreases it for others. DDT and methoxychlor are examples of insecticides that are less toxic at higher temperature; toxaphene, endrin, malathion, and parathion are more toxic (*32*). The effect can be very pronounced, as shown in Table 9. There are differences in response within species as well as between species. Diquat, for example, was toxic to female mosquitofish under conditions in which the males were not affected (*170*).

It is important to recognize that the high toxicity to fish of some herbicides (trifluralin, for example) is tempered in nature by processes that inactivate the compound, such as strong adsorption and relative immobility on soil surfaces, so that contamination of natural water bodies is minimal if the chemical is used according to label directions.

Another aspect of toxicity to fish is the development of resistance to pesticides, which has been documented for several species of fish. For example, fish in a contaminated lake in Mississippi had higher tolerance for endrin, DDT, and toxaphene than those in a relatively clean lake (*22*). Toxicity will also result in the flourishing of one group of organisms in an aquatic environment while others are suppressed. Thus, elimination of algae-eating species will produce increases in algae and anaerobic bacteria populations (*130*).

Chronic Toxicity

Fish mortalities have been observed to occur in nature by long-term, low-level exposure to pesticides. Numerous pathological effects on the tissues and organs of fish have also been noted, including lesions of liver and gills and changes in the intestines, kidneys, brain, and blood. However, some chemicals—notably methoxychlor and carbamate insecticides such as carbofuran and carbaryl—are rapidly hydrolyzed on ingestion and therefore do not have chronic effects.

Fish Reproduction and Growth

Persistent pesticides such as DDT and dieldrin can have strong adverse effects on reproduction in fish. In one typical case, the hatching success of landlocked Atlantic salmon from a lake contaminated with DDT was 36% lower than in control fish (*109*). Mortality usually occurs in salmon sac-fry during the period of

Table 9. Effect of water temperature and exposure time on the toxicity of trifluralin to bluegills[1]

Water temperature	48-Hour LC_{50}	24-Hour LC_{50}
°*F*	*μg/liter*	*μg/liter*
85	8.4	10
75	66	120
65	200	360
55	380	530
45	590	1300

[1] From Cope (48).

yolk-sac absorption. Some herbicides also may produce reproduction problems in fish, causing atrophy of spermatic tubules and production of abnormal spermatozoa (*48*). In addition to reproductive failures, loss of appetite and restricted growth have been reported to result from exposure of fish to pesticides.

Fish Behavior

Cases have been reported of changes in the conditioned responses and locomotor patterns of fish as a result of exposure to pesticides. One well-documented effect is an increase in sensitivity to low water temperatures, including active avoidance, in fish exposed to DDT (*32*).

Effects on Aquatic Plants

Herbicides, by their very nature, are more toxic to aquatic plants than insecticides, but adverse effects do not always occur where they might be expected. Phytoplankton are sometimes unaffected by pesticides, sometimes multiply when predators are removed by the chemicals, and sometimes are seriously inhibited in growth, depending on the particular conditions at hand. One apparent and obviously serious effect is the disruption of photosynthesis in phytoplankton: in a comprehensive series of tests, carbon fixation by estuarine phytoplankton was reduced by 45 of 54 chemicals tested, with reductions of over 90% for several of the compounds (*158*). Some aquatic plants act as concentrating agents for the organochlorine insecticides, so that when the plants die, concentrations of the pesticides, as well as of plant nutrients, are released into the water.

Odor and Taste

Several of the organochlorine insecticides, including toxaphene, endrin, and heptachlor, impart objectionable odors to water at concentrations of only a few parts per billion; a number of herbicides generate a strong odor; and the solvents used in many formulations are highly odorous in concentrations as low as 16 ppb (*131*). The herbicide 2,4-D hydrolyzes in water to 4-chlorophenol and 2,4-dichlorophenol, both of which impart displeasing flavors and odors at low ppb levels. However, the phenols are only rarely detected in natural waters (*72*). Decomposing aquatic plants that have been killed by herbicides are a major source of foul odors in aquatic environments.

REMOVAL OF PESTICIDES FROM THE AQUATIC ENVIRONMENT

Obviously, no removal of pesticide residues from bodies of water would be needed if chemicals that are rapidly degradable in the aquatic environment were the only ones being used, but such is not the case. Pesticides that are relatively persistent pose a threat to water quality that has prompted investigations of methods for their removal, chiefly in connection with the protection of drinking water supplies. Residues can be removed directly from water bodies by such measures as dredging sediments and removing weeds, debris, and coarse fish. However, these methods are generally not economically feasible and the method that is generally followed for nondrinking waters is to simply allow a period of time for natural renovation to occur.

With public water supplies, available data indicate that present-day conventional water-treatment processes, such as lime-alum coagulation, sedimentation, sand filtration, chlorination, and pH adjustment, will reduce high pesticide levels substantially, but are inadequate for removal of chronic contamination at low levels (*39*). The degree of removal depends on the water solubility and adsorptivity of the individual pesticides, with more efficient removal for compounds of low solubility or high adsorptivity. In one series of tests, for example, conventional treatment eliminated less than 10% of the lindane and only 20% of the parathion in the water, but removed 55% of the dieldrin, 63% of the 2,4,5-T ester, and 98% of the DDT (*135*).

Much research effort has been expended on adsorbents to purify the water beyond the levels attainable by conventional treatment. Activated carbon is clearly the most effective adsorbent for pesticides, having a removal efficiency about 4 orders of magnitude greater than that of soil, 3 orders greater than that of algae, and 2 orders greater than that of coal (*100*). The effectiveness of removal depends on the contact time, the concentration of activated carbon, the concentration of the pesticides, and the presence of organic material in the water that may compete with the pesticides for adsorption sites on the carbon. If the water is clarified by other processes before the activated carbon is introduced, the competitive organic matter can be at least partially controlled (*121*). Removal efficiency decreases at pesticide concentrations below 1 ppb. Although it is possible to remove lower concentrations of organochlorine insecticides, inordinately large amounts of carbon are required (*39*). At the 1-ppb level, organochlorine insecticides in drinking water account for only about 5% of the dietary intake of these pesticides and would not pose a threat to human health, at least with respect to acute effects (*94*).

Other techniques for pesticide removal from water have been explored, with limited success. Chemical or biological oxidation will degrade some pesticides, but toxic products are formed in many cases (*39*); ozone will attack even the stable organochlorine insecticides, but only at large and impractical concentrations and with unknown products (*135*); and strongly basic anion-exchange resins will remove high concentrations of such pesticides as 2,4-D salts (*3*). Use of reverse osmosis membranes has shown promise for removal of a wide variety of pesticides (*40*). Other effective processes may yet be developed that will take advantage of specific characteristics of individual pesticides, one possible example being the introduction of microorganisms that are specific for rapid inactivation of certain classes of chemicals.

PRACTICES FOR REDUCING ENTRY OF PESTICIDES INTO THE AQUATIC ENVIRONMENT

A total of 15 pesticide management practices, designated as P 1 through P 15, are presented in Volume I (Table 18 and Section 4.4). These practices reduce pesticide losses in runoff from treated fields by manipulation of the chemical itself and are meant to supplement the basic control of runoff and erosion carrying the pesticide, which is dealt with in other sections of Volume I. The various aspects of the practices are discussed in Volume I without supportive documentation. Appropriate documentation for each of the practices is, however, presented in Table 10.

Table 10 contains citations of articles supporting direct statements made in Volume I and of articles containing closely related information that may be of benefit in evaluating the individual practices. Information for both Volume I and this volume was obtained not only from the published papers cited, but also from discussions and meetings with agricultural and pesticide specialists and from internal progress reports of the CRIS (Current-Research-in-Science) information retrieval system of the U.S. Department of Agriculture and the State agricultural experiment stations.

RESEARCH NEEDS

The length of this review constitutes first-hand evidence that much is already known about pesticides in the aquatic environment, yet numerous important avenues for future research remain. Further efforts should, of course, be directed to minimization of those agricultural practices that contribute to erosion and runoff from cropland and thereby produce excessive losses of applied chemicals, but our concern here is with aspects of the system that deal with the pesticides directly. Such needs appear to fall into five general areas: (1) prediction of pesticide behavior in the aquatic ecosystem; (2) definition of significance of residues occurring in water bodies; (3) investigation of means for lowering rates and frequency of application of pesticides, so that the potential for contamination of waters would be lessened; (4) development of new pesticides having environmentally favorable properties; and (5) research on corrective measures to reduce or remove contamination by applied pesticides. No order of priority among these is intended in the brief discussions that follow.

Prediction of Pesticide Behavior

The development of mathematical models to predict the behavior of pesticides after application is a relatively new and important area of research. The ideal model is one that is able to predict the consequences of any given practice or set of conditions; applying it, one can identify optimum modes of pesticide use with respect to some particular attribute. However, a large amount of work is required to construct useful models of the typically complex agricultural systems. Moreover, data collection could be a limiting step; a complete model may demand such an elaborate input of hydrologic, chemical, biologic, and management data that collection of real-world numbers, including analyses, might take so long that events would outrun predictions.

Despite these difficulties, development and refinement of exploratory models is being actively pursued. With respect to pesticide movement in runoff, a model has been developed with the objective of minimizing water pollution (*12, 53*). It takes into account conditions both during and between runoff events, and has given satisfactory predictions for pesticides moved entirely on sediment, but not yet for those moved in both water and sediment. Models in related areas have also been proposed. The movement of agricultural chemicals through the soil profile has been described mathematically, but existing models do not take into account the ongoing natural soil-forming processes, so that their

Table 10. Bibliography on pesticide management practices (Volume I, Section 4.4)

Pesticide management practice			Citations	Significant subjects
No.	Page No. in Vol. I.	Description		
P1	85	Using Alternative Pesticides	Craig *et al* (52)	Typical examples of alternative pesticides effective against same pests in same crops
			Bailey (10)	Discussion of pesticide properties pertinent to movement in runoff
			Spencer (144)	Discussion of pesticide properties pertinent to movement in runoff
			Erbach and Lovely (64)	Desirability of rotating equally effective pesticides in succeeding years on same crop
P2	85	Optimizing Pesticide Placement With Respect to Loss	Caro *et al* (35)	Comparative pesticide loss in runoff: in-furrow vs. broadcast applications
			Ritter *et al* (134)	Comparative pesticide loss in runoff: ridge planting vs. surface-contour planting
			Apple (7)	Comparative toxicity to crop seed of insecticides when placed in seed furrow
			Constien *et al* (44)	Necessity for placing insecticides in seed furrow in no-till management
			Moomaw and Robison (116)	Satisfactory performance of herbicides placed in narrow bands
			Reid and Peacock (132)	Subsurface sweep applicators
			Erbach, Lovely, and Bockhop (65)	High efficiency of precise spacing of herbicides
			Bode and Gebhardt (25)	Advantage of disk over other implements in incorporating pesticides to minimize loss in runoff
			Wax (159)	Current trends toward broadcast application
P3	85	Using Crop Rotation	Epstein and Grant (63)	Lower runoff loss of pesticides from rotation
			Wax (159)	Weed reduction by crop rotation
			Stockdale, DeWitt, and Ryan (147)	Improved insect control by crop rotation
			Daniels (55)	Improved insect control by crop rotation
			Fleming (69), p. 327	Typical examples of improved insect control by crop rotation
			Blakely, Coyle, and Steele (23), p. 305	Reduction in erosional losses by rotation
			Wade (155)	Intercropping of corn and peanuts to reduce attack by corn borers

Table 10. (continued)

Pesticide Management Practice			Citations	Significant subjects
No.	Page No. in Vol. I.	Description		
			Kuhlman, Cooley, and Walt (103)	Comparative acreage receiving insecticides: continuous crop vs. rotations
			Allaway (1), p. 391-2	Comparative weed control: continuous crop vs. rotations
P4	86	Using Resistant Crop Varieties	Sprague and Dahms (146)	Review of crop resistance to insects and reductions in use of insecticides
			Chant (38), p. 204	Wheat varieties resistant to Hessian fly
			Hoffman (93)	Examples of resistant crop varieties and insects resisted
P5	86	Optimizing Crop Planting Time	Fleming (69), p. 328	Time of crop planting: summary of effects on insect infestations
			Craig *et al* (52), p. 48	Advantages of early plantings for combatting European corn borer
			Wellhausen (164)	Break in sorghum plantings to combat sorghum midge
			Wax *et al* (160)	Advantages of late plantings of soybeans to combat weeds
			Burkhead *et al* (28)	Summary of planting dates of field crops
P6	86	Optimizing Pesticide Formulation	Wax (159)	Addition of surfactants to increase penetration of herbicides; comparability of liquids and granules
			Foy and Bingham (70)	Surfactants or oils to enhance herbicide penetration in plants
			Mullison (119)	Toxicity of components of formulations other than active ingredient
			Barnett *et al* (16)	Comparative runoff potential: 2, 4-D esters vs. amine salt
			Miles and Woehst (115)	Controlled release formulations
			Clack (42)	Use of foam formulations for weed control
			Depew (57)	Superiority of granular formulations over liquids in seed-furrow applications
P7	87	Using Mechanical Control Methods	Burnside and Colville (29)	Superiority of tillage-herbicide combination for weed control
			Wax (159)	Discussion of tillage and flame cultivation for weed control
			Whitaker, Heinemann, and Wischmeier (165)	Ability of cultivation to reduce erosive soil loss
			Crafts and Robbins (51), p. 140-154	Discussion of tillage methods in weed control
			Behrens (20)	Disadvantages of cultivation and tillage

Table 10. (continued)

Pesticide management practice			Citations	Significant subjects
No.	Page No. in Vol. I.	Description		
P8	87	Eliminating Excessive Treatment	Turnipseed *et al* (152)	Effectiveness of lower than recommended rate for insect control in soybeans
			Chiang (41)	Development of recommendations for determining threshold pest damage
			Shore (143)	Mathematical computation of optimum dosages
P9	87	Optimizing Time of Day for Pesticide Application	Ware *et al* (157)	Higher efficiency of early morning spraying
			Cooperative Exten. Serv., Illinois (47), p. 238	Timing of spray to avoid harm to honeybees
P10	87	Optimizing Date of Pesticide Application	Apple, Walgenbach, and Knee (8)	Comparative effectiveness: planting-time vs. cultivation-time treatments for corn rootworm
			Harrison and Press (86)	Timing of sprays against corn borer
			Anderson (4)	Optimization of time of foliar application of herbicides
			Texas Agr. Exten. Serv. (149)	Recommendations for long-interval preplant applications of herbicides on cotton and peanuts
			Erbach and Lovely (64)	Critical period for applying herbicides on corn and soybeans
			Summers, Byrne, and Pimentel (148)	Advantages of early insecticide application for alfalfa weevil control
P11	88	Using Integrated Control Programs	Hanson (84)	Aspects of integrated pest control programs
			Council Environ. Qual. (49)	Aspects of integrated pest control programs
			Chant (38)	Aspects of integrated pest control programs
			Casey, Lacewell, and Sterling (37)	Example of reduction in insecticide use by introduction of pest management strategy
			Giese, Peart, and Huber (73)	Reliability of computer-based pest management systems
P12	88	Using Biological Control Methods	Knipling (101)	Overview of biological control of insects
			Quraishi (129)	Aspects of biological control of insects
			Putnam and Duke (128)	Example of biological control of weeds
			Patti and Carner (124)	Example of usefulness of Bacillus thuringiensis
P13	88	Using Lower Pesticide Application Rates	Turnipseed *et al* (152)	Examples of effectiveness of rates less than recommended levels
			Casey, Lacewell, and Sterling (37)	Example of lower dosage in an integrated control program

Table 10. (continued)

Pesticide Management Practice			Citations	Significant subjects
No.	Page No. in Vol. I.	Description		
			Johnstone (96)	Insecticide application in ultra-low-volume sprays
P14	89	Managing Aerial Applications	Wax (159)	Advantages and disadvantages of aerial application of herbicides
			Marston *et al* (111)	Pesticide in stream water from aerial spraying
			Cole, Barry, and Frear (43)	DDT in environment after aerial application
			Glotfelty and Caro (74)	Movement of airborne pesticides
P15	89	Planting Between Rows in Minimum Tillage	Coop. Exten. Serv. Ohio (46)	Practice recommended for reduction of corn rootworm populations
			Way (161)	Disadvantages of practice

ability to predict field behavior is limited (*24*). A relatively simple model has been developed (*79*) in which the quantities of pesticide to be applied are optimized with respect to profits to the grower. Refinement of this model could well lead to reduced use of pesticides and lessened environmental contamination.

Significance of Residues

The true significance of pesticide residues in the environment is perhaps the least understood aspect of the system, particularly with regard to chronic contamination at very low concentrations. Changes in behavioral patterns of aquatic organisms have been observed as a result of chronic exposure, but little is truly known of possible long-term, subtle effects (*118*). To aid in assessment of the hazard, we need (1) in-depth studies of declining species; (2) studies of the gain, loss, or change in residues in both living and nonliving components of the environment, to relate trends to observable effects; and (3) toxicological measurements under conditions that simulate the natural environment more closely than the conditions used in such tests in the past.

Reducing Pesticide Use

Research directed to reduction in use of pesticides by more efficient application of the chemical or by substitution of nonchemical methods of pest control offers the broadest opportunity for decreasing the potential for environmental contamination by pesticides. One important facet that will require considerable future effort is the development of large-scale integrated control programs in which minimum amounts of chemical pesticides are used. Much more information is needed for integrated control than is generally required to use pesticides alone. A successful program for insect control, for example, requires knowledge of the dynamics of the pest population, life history of the pest, natural enemies, nutritional requirements, host plants, economic threshold of the insect population, and behavior of chemicals and organisms used in the program with respect to effects on nontarget environmental components (*93, 114*). Each of these must be examined in detail and interrelated and there are also researchable associated matters, such as the selection of the most suitable of alternative methods of control for incorporation into a program, the use of adverse natural phenomena to signal the appropriate time for attacking insects by integrated control, and the bringing of experimentally proven integrated control methods up to practical application (*38, 93*).

Several approaches to more efficient application and utilization of pesticides are being actively investigated, but require additional effort. One such is the development and testing of foams, gels, and polymer-encapsulated slow-release formulations that can reduce drift, minimize movement of the pesticide in the environment, and perhaps decrease the number of applications needed because the chemical will be used more efficiently. A second is the use of electrostatically charged sprays to decrease drift and optimize deposition of the chemicals onto plant surfaces. Optimization of spray droplet particle sizes for efficient on-target deposition is also being investigated. A third avenue under investigation is the reduction of pesticide volatilization from plant and soil surfaces, and a fourth is the precision placement of pesticides in the soil by devices such as subsurface sweep applicators. Another aspect worthy of further study is the development of computer programs to predict occurrences of pest infestations. Such a program has already been successful with potato late blight, saving growers an average of 4 sprays annually in comparison with the normal practice of spraying at 10-day intervals (*102*).

Development of New Pesticides

Present pesticides, although effective, are far from perfect. Opportunity is still great to develop new chemicals that are highly specific for the pest, safe to man and wildlife, and have little effect on the quality of the environment. The ideal compound would have low solubility in fats to minimize the possibility of biomagnification; would be biodegradable, but only after its intended function is completed; and would be nontoxic, but convertible to a toxicant in the presence of the pest (*113*).

Corrective Measures

Research should be conducted on means for shortening the persistence of relatively stable compounds at the site of application or in the aquatic environment by deliberate manipulation of their modes of dissipation. Work of this type would be directed to such goals as enhancement of volatility and photodecomposition or adjustment of adsorptivity and leachability, perhaps by use of adjuvants. Other possible corrective efforts could involve the use of aquatic plants as traps to remove residues from water by absorption, addition of an inoculum of microorganisms to degrade the pesticides after their biocidal activity is no longer needed, direct chemical inactivation in soil by addition of a reactant, use of additives to control the metabolic degradation of herbicides within plant tissues, and use of new adsorptive media or ion exchange resins to remove residues from water (*70*).

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CHAPTER 6

INTERDISCIPLINARY RESEARCH NEEDS

B. A. Stewart

The preceding chapters have listed research needs relative to particular disciplines. This chapter briefly discusses their interrelationships as they relate to nonpoint pollution from cropland. Nonpoint pollution can best be controlled by controlling erosion. However, knowledge is critically lacking concerning the degree of erosion and sediment control necessary to control nutrient and pesticide losses.

The Universal Soil Loss Equation is being used extensively in predicting nonpoint pollution. It is an excellent tool because it quantifies soil loss and the effect of various practices on reducing this loss. The Universal Soil Loss Equation, however, estimates the average annual soil loss from a field area and does not indicate the amount of sediment that is delivered to a stream. The sediment delivery ratio is an attempt to relate field losses to amounts reaching the stream. The difficulty with this concept for predicting water pollution is that it takes into account both the deposition of sediment as it moves toward the stream and the gains from channel erosion. Pollutants, particularly pesticides and nutrients from added fertilizers, are usually not associated with sediment from channel erosion.

The enrichment ratio is a measure of the increase in the concentration of a pollutant associated with the sediment that actually reaches a stream compared to the concentration in the watershed soil. The concentration usually increases because more nutrients and pesticides are adsorbed on fine-textured particles than on coarse particles, and more coarse particles are deposited as the sediment moves from the field area to the stream. Consequently, the most pressing research need is to gain a better understanding of how the Universal Soil Loss Equation, the sediment delivery ratio, and the enrichment ratio can be meshed. These measurements and evaluations will not be made easily, cheaply, or quickly. Data obtained from small plots or field area can provide only crude applications because the size effect is so significant. Since cropland is so diffuse, monitoring of agricultural areas is impractical. The most likely approach, therefore, is to develop predictive models. A major effort should be directed toward obtaining necessary data and developing such models. Results from such an effort cannot be expected to be precise but should represent a statistical approach that establishes relationships and relative levels.

The Universal Soil Loss Equation predicts average annual soil losses. Additional accuracy is needed to predict single events. These can be important in their effect on water quality, especially if the time of loss is associated with applications of agricultural chemicals. Again, information is needed on large watershed areas. It is fairly easy to instrument a field-sized area and measure losses of sediment and associated pollutants. The difficulty is in determining how much of these actually reach the stream. The effects and proper design of filter strips, settling basins, and sediment traps should also be determined. Some evidence indicates that these can very effectively reduce losses; however, there are also indications that this deposited material may be moved during extreme runoff events.

No-till and conservation tillage systems are highly effective for controlling soil loss. Additional research is needed, however, because it is not known how widely these practices can be used. Insect and disease hazards are greater, and if these practices are used on vast acreages, outbreaks could occur. Also, in some areas, it may be necessary to occasionally plow to loosen the soil. More data are also required concerning pesticide and nutrient losses from no-till and conservation tillage systems as compared to conventional systems. Sediment losses are drastically reduced, but the meager data available do not show proportionate losses in pollutants.

Another primary research need is to evaluate the economic impact of control measures for controlling sediment and chemical losses. Available technology is adequate to control these pollutants in most instances. Implementation of these measures could produce major changes in cultivation practices, timing of chemical applications, and location of production and general farming practices. These changes would have direct effects on the profitability of agricultural production, supplies of food and fiber, and rural land use. An urgent need is to appraise alternative procedures for controlling pollutants so that economic hardships can be minimized.

Since our landscapes are not uniform and all areas are not equidistant from a water body, numerous combinations of practices can achieve the same water-quality goal. The most desirable approach is to give each farmer the opportunity to select the appropriate combination of practices. To do this, he needs to know how much control each practice will provide. The costs and benefits of implementing the various practices should also be known. This should include regional and national impacts. Thus, much research is needed in a variety of disciplines and the first step in any of them is to quantify the relationships.

Last, and perhaps most important, criteria must be established as to what levels of sediment and chemicals constitute pollution. The criteria might be absolute limits (not to exceed X ppm) or conditional limits (not to exceed Y ppm in Z years). Although the establishment of these criteria will require research outside of traditional agricultural areas–limnology, aquatic botany and zoology, and water treatment–it is of great importance to the agricultural community. Without these criteria it is impossible to determine which practice will be adequate, much less decide what the economic impacts will be.

APPENDIX A

SIMULATION OF DAILY POTENTIAL DIRECT RUNOFF

INTRODUCTION

The amount and seasonal distribution of direct runoff was estimated to assess potential transport of pesticides and nutrients. The effects of some land management practices on direct runoff were also estimated. Hydrologists have developed several rainfall-runoff models of various degrees of complexity for making these estimates. The more physically realistic models are quite complicated and require a great deal of input information and computer time. The national scope of this report and the severe time constraints involved dictated the use of a rather simple method of estimating runoff from rainfall. Any input information required must also be readily available. After considering several possibilities, we decided to use the Soil Conservation Service procedure for estimating direct runoff from storm rainfall (*4*).

THE SOIL CONSERVATION SERVICE PROCEDURE FOR ESTIMATING DIRECT RUNOFF FROM STORM RAINFALL

The Soil Conservation Service procedure for estimating direct runoff from storm rainfall (sometimes called the SCS curve number method) was designed to use the most generally available rainfall data: total daily rainfall. For this reason rainfall intensity is largely ignored. The basic relationship is the equation:

$$Q = \frac{(P-I_a)^2}{(P-I_a)+S} \;;\; P \geq I_a \qquad (1)$$

where

- Q = runoff in inches
- P = rainfall in inches
- I_a = initial abstraction in inches
- S = potential maximum retention plus initial abstraction.

The initial abstraction before runoff begins is considered to consist mainly of interception, infiltration and surface storage. Utilizing limited data from small experimental watersheds, the following empirical relationship was developed:

$$I_a = (0.2)S. \qquad (2)$$

Substituting this relationship into equation (1) gives

$$Q = \frac{(P-0.2S)^2}{P+0.8S} \;,\; P \geq (0.2)\,S\;, \qquad (3)$$

which is the rainfall-runoff relation used in the SCS method.

The parameter CN (runoff curve number of hydrologic soil-cover complex number) is defined in terms of the parameter S as:

$$CN = \frac{1000}{S+10} \qquad (4)$$

Note that runoff equals rainfall when S = 0 and CN = 100.

The potential maximum retention, S , and therefore the runoff curve number are related to soil surface and profile properties, the vegetative cover, management practices, and the soil water content on the day of the storm. Solutions of equation (3) are shown as a family of curves in Fig. 1.

Soil water content on the day of the storm is accounted for by an Antecedent Moisture Condition (AMC) determined by the total rainfall in the 5-day period preceding the storm.

Three AMC groups have been established with the boundaries between groups dependent upon the time of year as shown in Table 1.

The seasonal difference in the AMC groupings is an attempt to account for the greater evapotranspiration between storms during the growing season.

The different infiltration characteristics of soils are accounted for by classifying soils into four groups based

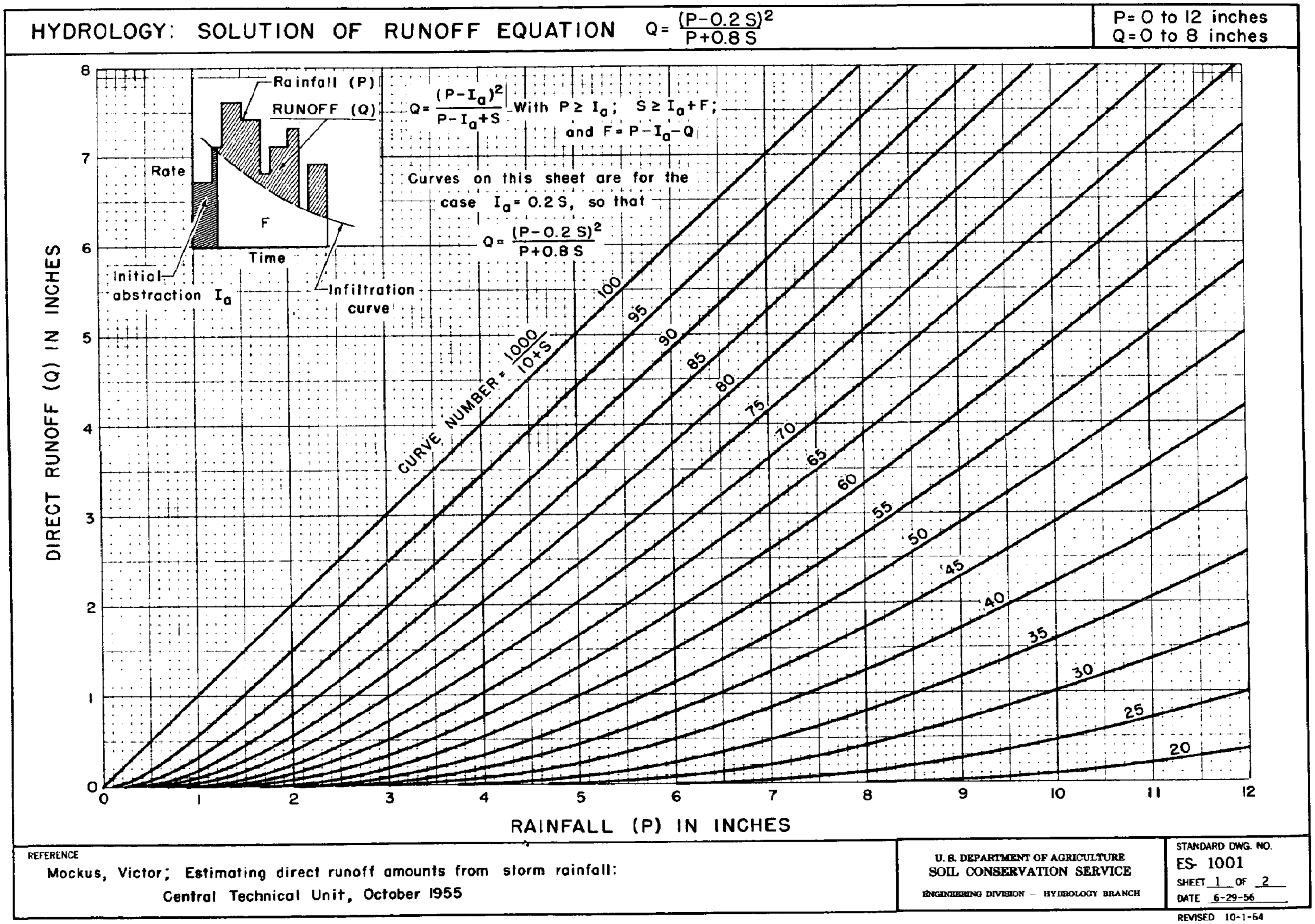

Figure 1.–Solutions of Eq. 3. [From SCS National Engineering Handbook (4)]

Table 1. Seasonal rainfall limits for antecedent moisture conditions[1]

AMC group	Total 5-day antecedent rainfall	
	Dormant season	Growing season
	inches	*inches*
I	<0.5	<1.4
II	0.5 - 1.1	1.4 - 2.1
III	>1.1	>2.1

[1] From SCS National Engineering Handbook (4).

upon the minimum rate of infiltration obtained for a bare soil after prolonged wetting. The influences of both the surface and the profile of a soil are included. The hydrologic soil groups as defined by SCS soil scientists in the National Engineering Handbook are:

A. (Low runoff potential). Soils having high infiltration rates even when thoroughly wetted and consisting chiefly of deep, well to excessively drained sands or gravels. These soils have a high rate of water transmission.

B. Soils having moderate infiltration rates when thoroughly wetted and consisting chiefly of moderately deep to deep, moderately well to well drained soils with moderately fine to moderately coarse textures. These soils have a moderate rate of water transmission.

C. Soils having slow infiltration rates when thoroughly wetted and consisting chiefly of soils with a layer that impedes downward movement of water, or soils with moderately fine to fine texture. These soils have a slow rate of water transmission.

D. (High runoff potential). Soils having very slow infiltration rates when thoroughly wetted and consisting chiefly of clay soils with a high swelling potential, soils with a permanent high water table, soils with a claypan or clay layer at or near the surface, and shallow soils over nearly impervious material. These soils have a very slow rate of water transmission.

The SCS has classified over 9,000 soils in the United States and Puerto Rico according to the above scheme. A sample from the extensive table in the SCS National Engineering Handbook is shown in Table 2. Rainfall-runoff data from small watersheds or infiltrometer plots were used to make the classifications where such data were available, but most are based on the judgement of soil scientists and correlators who used physical properties of the soils in making the assignments.

The interaction of hydrologic soil group (soil) and land use and treatment (cover) is accounted for by assigning a runoff curve number for average soil moisture condition (AMC II) to important soil cover complexes for the fallow period and the growing season. Rainfall-runoff data for single soil cover complex watersheds and plots were analyzed to provide a basis for making these assignments. Average runoff curve numbers for several soil-cover complexes are shown in Table 3. Average runoff curve numbers (AMC II) are for the average soil moisture conditions. AMC I has the lowest runoff potential. AMC III has the highest runoff potential. Under this condition the watershed is practically saturated from antecedent rains. Appropriate curve numbers for AMC I and III based upon the curve number for AMC II are shown in Table 4.

Curve numbers for a "good hydrologic condition" were used in the potential direct runoff simulations. "Hydrologic condition" refers to the runoff potential of a particular cropping practice. A row crop in good hydrologic condition will have higher infiltration rates and, consequently, less direct runoff than the same crop in poor hydrologic condition. Good hydrologic condition seemed an appropriate description of corn under modern management practices.

Seasonal variation not accounted for by the seasonal dependency of the AMC classes is included by varying the average moisture condition curve number according to the stages of growth of a particular crop. For the simulations reported here, with straight row corn as the index crop, the average (AMC II) curve number was set equal to that for fallow for the period from March 1 until the average emergence date for corn. Emergence dates were assumed to be 2 weeks after the average planting date reported by the USDA (*5*). During the growing season, AMC II curve numbers for each day were calculated by the following equation:

$$CN_i = F - \frac{C_i}{C_{ave}} (F - CN_{ave}) \quad . \tag{5}$$

where

CN_i = the curve number for the ith day for AMC II.

F = fallow curve number.

C_i = crop coefficient for the ith day. $C_i \leq 1$.

C_{ave} = average crop coefficient for the growing season.

CN_{ave} = average growing season curve number for AMC II.

The crop coefficients C_i are defined as the ratio of the crop evapotranspiration to potential evapotranspiration for a given day when soil water is not limiting. Crop

Table 2.–Soil names and hydrologic classifications[1] (Sample)

AABERG	C	AHL	C	ALMY	B	ANLAUF	C	AROOSTOOK	
AASTAD	B	AHLSTROM	C	ALOHA	C	ANNABELLA	B	AROSA	C
ABAC	D	AHMEEK	B	ALONSO	B	ANNANDALE	C	ARP	C
ABAJO	C	AHOLT	D	ALOVAR	C	ANNISTON	B	ARRINGTON	B
ABBOTT	D	AHTANUM	C	ALPENA	B	ANOKA	A	ARRITOLA	D
ABBOTTSTOWN	C	AHWAHNEE	C	ALPHA	C	ANONES	C	ARROLIME	C
ABCAL	D	AIBONITO	C	ALPON	B	ANSARI	D	ARRON	D
ABEGG	B	AIKEN	B/C	ALPOWA	B	ANSEL	B	ARROW	B
ABELA	B	AIKMAN	D	ALPS	C	ANSELMO	A	ARROWSMITH	B
ABELL	B	AILEY	B	ALSEA	B	ANSON	B	ARROYO SECO	B
ABERDEEN	D	AINAKEA	B	ALSPAUGH	C	ANTELOPE SPRINGS	C	ARTA	C
ABES	D	AIRMONT	C	ALSTAD	B	ANTERO	C	ARTOIS	C
ABILENE	C	AIROTSA	B	ALSTOWN	B	ANT FLAT	C	ARVADA	D
ABINGTON	B	AIRPORT	D	ALTAMONT	D	ANTHO	B	ARVANA	C
ABIQUA	C	AITS	B	ALTAVISTA	C	ANTHONY	B	ARVESON	D
ABO	B/C	AJO	C	ALTDORF	D	ANTIGO	B	ARVILLA	B
ABOR	D	AKAKA	A	ALTMAR	B	ANTILON	B	ARZELL	C
ABRA	C	AKASKA	B	ALTO	C	ANTIOCH	D	ASA	B
ABRAHAM	B	AKELA	C	ALTUGA	C	ANTLER	C	ASBURY	B
ABSAROKEE	C	ALADDIN	B	ALTON	B	ANTOINE	C	ASCALON	B
ABSCOTA	B	ALAE	A	ALTUS	B	ANTROBUS	B	ASCHOFF	B
ABSHER	D	ALAELOA	B	ALTVAN	B	ANTY	B	ASHBY	C
ABSTED	D	ALAGA	A	ALUM	B	ANVIK	B	ASHCROFT	B
ACACIO	C	ALAKAI	D	ALUSA	D	ANWAY	B	ASHDALE	B
ACADEMY	C	ALAMA	B	ALVIN	B	ANZA	B	ASHE	B
ACADIA	D	ALAMANCE	B	ALVIRA	C	ANZIANO	C	ASHKUM	C
ACANA	D	ALAMO	D	ALVISO	D	APACHE	D	ASHLAR	B
ACASCO	D	ALAMOSA	C	ALVOR	C	APAKUIE	A	ASHLEY	A
ACEITUNAS	B	ALAPAHA	D	AMADOR	D	APISHAPA	C	ASH SPRINGS	C
ACEL	D	ALAPAI	A	AMAGON	D	APISON	B	ASHTON	B
ACKER	B	ALBAN	B	AMALU	D	APOPKA	A	ASHUE	B
ACKMEN	B	ALBANO	D	AMANA	B	APPIAN	C	ASHUELOT	C
ACME	C	ALBANY	C	AMARGOSA	D	APPLEGATE	C	ASHWOOD	C
ACO	B	ALBATON	D	AMARILLO	B	APPLETON	C	ASKEW	C
ACOLITA	B	ALBEE	C	AMASA	B	APPLING	B	ASO	C
ACOMA	C	ALBEMARLE	B	AMBERSON		APRON	B	ASOTIN	C
ACOVE	C	ALBERTVILLE	C	AMBOY	C	APT	C	ASPEN	B
ACREE	C	ALBIA	C	AMBRAW	C	APTAKISIC	B	ASPERMONT	B
ACRELANE	C	ALBION	B	AMEDEE	A	ARABY		ASSINNIBOINE	B
ACTON	B	ALBRIGHTS	C	AMELIA	B	ARADA	C	ASSUMPTION	B
ACUFF	B	ALCALDE	C	AMENIA	B	ARANSAS	D	ASTATULA	A
ACWORTH	B	ALCESTER	B	AMERICUS	A	ARAPIEN	C	ASTOR	A/D
ACY	C	ALCOA	B	AMES	C	ARAVE	D	ASTORIA	B
ADA	B	ALCONA	B	AMESHA	B	ARAVETON	B	ATASCADERO	C
ADAIR	D	ALCOVA	B	AMHERST	C	ARBELA	C	ATASCOSA	D
ADAMS	A	ALDA	C	AMITY	C	ARBONE	B	ATCO	B
ADAMSON	B	ALDAX	D	AMMON	B	ARBOR	B	ATENCIO	B
ADAMSTOWN		ALDEN	D	AMOLE	C	ARBUCKLE	B	ATEPIC	D
ADAMSVILLE	C	ALDER	B	AMOR	B	ARCATA	B	ATHELWOLD	B
ADATON	D	ALDERDALE	C	AMOS	C	ARCH	B	ATHENA	B
ADAVEN	D	ALDERWOOD	C	AMSDEN	B	ARCHABAL	B	ATHENS	B
ADDIELOU	C	ALDINO	C	AMSTERDAM	B	ARCHER	C	ATHERLY	B
ADDISON	D	ALDWELL	C	AMTOFT	D	ARCHIN	C	ATHERTON	B/D
ADDY	C	ALEKNAGIK	B	AMY	D	ARCO	B	ATHMAR	C
ADE	A	ALEMEDA	C	ANACAPA	B	ARCOLA	C	ATHOL	B
ADEL	A	ALEX	B	ANAHUAC	D	ARD	C	ATKINSON	B
ADELAIDE	D	ALEXANDRIA	C	ANAMITE	D	ARDEN	B	ATLAS	D
ADELANTO	B	ALEXIS	B	ANAPRA	B	ARDENVOIR	B	ATLEE	C
ADELINO	B	ALFORD	B	ANASAZI	B	ARDILLA	C	ATMORE	B/D
ADELPHIA	C	ALGANSEE	B	ANATONE	D	AREDALE	B	ATOKA	C
ADENA	C	ALGERITA	B	ANAVERDE	B	ARENA	C	ATON	B
ADGER	D	ALGIERS	C/D	ANAWALT	D	ARENALES	A	ATRYPA	C
ADILIS	A	ALGOMA	B/D	ANCHO	B	ARENDTSVILLE	B	ATSION	C
ADIRONDACK		ALHAMBRA	B	ANCHORAGE	A	ARENOSA	A	ATTERBERRY	B
ADIV	B	ALICE	A	ANCHOR BAY	D	ARENZVILLE	B	ATTEWAN	A
ADJUNTAS	C	ALICEL	B	ANCHOR POINT	D	ARGONAUT	D	ATTICA	B
ADKINS	B	ALICIA	B	ANCLOTE	D	ARGUELLO	B	ATTLEBORO	
ADLER	C	ALIDA	B	ANCO	C	ARGYLE	B	ATWATER	B
ADOLPH	D	ALIKCHI	B	ANDERLY	C	ARIEL	C	ATWELL	C/D
ADRIAN	A/D	ALINE	A	ANDERS	C	ARIZO	A	ATWOOD	B
AENEAS	B	ALKO	D	ANDERSON	B	ARKABUTLA	C	AUBBEENAUBBEE	B
AETNA	B	ALLAGASH	B	ANDES	C	ARKPORT	B	AUBERRY	B
AFTON	D	ALLARD	B	ANDORINIA	C	ARLAND	B	AUBURN	C/D
AGAR	B	ALLEGHENY	B	ANDOVER	D	ARLE	B	AUBURNDALE	D
AGASSIZ	D	ALLEMANDS	D	ANDREEN	B	ARLING	D	AUDIAN	B
AGATE	D	ALLEN	B	ANDREESON	C	ARLINGTON	C	AU GRES	C
AGAWAM	B	ALLENDALE	C	ANDRES	B	ARLOVAL	C	AUGSBURG	B
AGENCY	C	ALLENS PARK	B	ANDREWS	C	ARMAGH	D	AUGUSTA	C
AGER	D	ALLENSVILLE	C	ANED	D	ARMIJO	D	AULD	D
AGNER	B	ALLENTINE	D	ANETH	A	ARMINGTON	D	AURA	B
AGNEW	B/C	ALLENWOOD	B	ANGELICA	D	ARMO	B	AURORA	C
AGNOS	B	ALLESSIO	B	ANGELINA	B/D	ARMOUR	B	AUSTIN	C
AGUA	B	ALLEY	C	ANGELO	C	ARMSTER	C	AUSTWELL	D
AGUADILLA	A	ALLIANCE	B	ANGIE	C	ARMSTRONG	D	AUXVASSE	D
AGUA DULCE	C	ALLIGATOR	D	ANGLE	A	ARMUCHEE	D	AUZQUI	B
AGUA FRIA	B	ALLIS	D	ANGLEN	B	ARNEGARD	B	AVA	C
AGUALT	B	ALLISON	C	ANGOLA	C	ARNHART	C	AVALANCHE	B
AGUEDA	B	ALLOUEZ	C	ANGOSTURA	B	ARNHEIM	C	AVALON	B
AGUILITA	B	ALLOWAY		ANHALT	D	ARNO	D	AVERY	B
AGUIRRE	D	ALMAC	B	ANIAK	D	ARNOLD	B	AVON	C
AGUSTIN	B	ALMENA	C	ANITA	D	ARNOT	C/D	AVONBURG	D
AHATONE	D	ALMONT	D	ANKENY	A	ARNY	A	AVONDALE	E

NOTES A BLANK HYDROLOGIC SOIL GROUP INDICATES THE SOIL GROUP HAS NOT BEEN DETERMINED
TWO SOIL GROUPS SUCH AS B/C INDICATES THE DRAINED/UNDRAINED SITUATION

[1] From SCS National Engineering Handbook (4).

Table 3.—Runoff curve numbers for hydrologic soil-cover complexes[1]

(Antecedent moisture condition II, and $I_a = 0.2\ S$)

Cover			Hydrologic soil group			
Land use	Treatment or practice	Hydrologic condition	A	B	C	D
Fallow	Straight row	----	77	86	91	94
Row crops	"	Poor	72	81	88	91
	"	Good	67	78	85	89
	Contoured	Poor	70	79	84	88
	"	Good	65	75	82	86
	" and terraced	Poor	66	74	80	82
	" " "	Good	62	71	78	81
Small grain	Straight row	Poor	65	76	84	88
	"	Good	63	75	83	87
	Contoured	Poor	63	74	82	85
	"	Good	61	73	81	84
	" and terraced	Poor	61	72	79	82
	" " "	Good	59	70	78	81
Close-seeded legumes[2] or rotation meadow	Straight row	Poor	66	77	85	89
	" "	Good	58	72	81	85
	Contoured	Poor	64	75	83	85
	"	Good	55	69	78	83
	" and terraced	Poor	63	73	80	83
	" " "	Good	51	67	76	80
Pasture or range		Poor	68	79	86	89
		Fair	49	69	79	84
		Good	39	61	74	80
	Contoured	Poor	47	67	81	88
	"	Fair	25	59	75	83
	"	Good	6	35	70	79
Meadow		Good	30	58	71	78
Woods		Poor	45	66	77	83
		Fair	36	60	73	79
		Good	25	55	70	77
Farmsteads		----	59	74	82	86
Roads (dirt)[3]		----	72	82	87	89
(hard surface)[3]		----	74	84	90	92

[1] From SCS National Engineering Handbook (4).
[2] Close-drilled or broadcast.
[3] Including right-of-way.

Table 4.–Curve numbers (CN) and constants for the case $I_a = 0.2S$[1]

1	2	3	4	5
CN for condition II	CN for conditions I	CN for conditions III	S values[2]	Curve[2] starts where P =
			(inches)	*(inches)*
100	100	100	0	0
99	97	100	.101	.02
98	94	99	.204	.04
97	91	99	.309	.06
96	89	99	.417	.08
95	87	98	.526	.11
94	85	98	.638	.13
93	83	98	.753	.15
92	81	97	.870	.17
91	80	97	.989	.20
90	78	96	1.11	.22
89	76	96	1.24	.25
88	75	95	1.36	.27
87	73	95	1.49	.30
86	72	94	1.63	.33
85	70	94	1.76	.35
84	68	93	1.90	.38
83	67	93	2.05	.41
82	66	92	2.20	.44
81	64	92	2.34	.47
80	63	91	2.50	.50
79	62	91	2.66	.53
78	60	90	2.82	.56
77	59	89	2.99	.60
76	58	89	3.16	.63
75	57	88	3.33	.67
74	55	88	3.51	.70
73	54	87	3.70	.74
72	53	86	3.89	.78
71	52	86	4.08	.82
70	51	85	4.28	.86
69	50	84	4.49	.90
68	48	84	4.70	.94
67	47	83	4.92	.98
66	46	82	5.15	1.03
65	45	82	5.38	1.08
64	44	81	5.62	1.12
63	43	80	5.87	1.17
62	42	79	6.13	1.23
61	41	78	6.39	1.28
60	40	78	6.67	1.33
59	39	77	6.95	1.39
58	38	76	7.24	1.45
57	37	75	7.54	1.51
56	36	75	7.86	1.57
55	35	74	8.18	1.64
54	34	73	8.52	1.70
53	33	72	8.87	1.77
52	32	71	9.23	1.85
51	31	70	9.61	1.92
50	31	70	10.0	2.00
49	30	69	10.4	2.08
48	29	68	10.8	2.16
47	28	67	11.3	2.26
46	27	66	11.7	2.34
45	26	65	12.2	2.44
44	25	64	12.7	2.54
43	25	63	13.2	2.64
42	24	62	13.8	2.76
41	23	61	14.4	2.88
40	22	60	15.0	3.00
39	21	59	15.6	3.12
38	21	58	16.3	3.26
37	20	57	17.0	3.40
36	19	56	17.8	3.56
35	18	55	18.6	3.72
34	18	54	19.4	3.88
33	17	53	20.3	4.06
32	16	52	21.2	4.24
31	16	51	22.2	4.44
30	15	50	23.3	4.66
25	12	43	30.0	6.00
20	9	37	40.0	8.00
15	6	30	56.7	11.34
10	4	22	90.0	18.00
5	2	13	190.0	38.00
0	0	0	infinity	infinity

[1] From SCS National Engineering Handbook (4).
[2] For CN in Column 1.

coefficient curves for corn were obtained by fitting a Fourier Series to a curve presented by Kincaid and Heermann (*2*).

Curve numbers for antecedent moisture conditions I and III were obtained from Table 4.

At harvesting date or when $CN_i = CN_{ave}$, whichever came first, the curve number was set equal to CN_{ave} and remained a constant until the next March 1. The SCS recommends that the after-harvest curve number be set equal to the average growing season curve number if 1/3 of the soil surface is exposed. This simulation represents a situation where residues are left on the field after harvest.

Any precipitation that occurred when the mean air temperature was less than 0°C was assumed to be snow and was accumulated as snow storage until the temperature went above 0°C. Snowmelt was calculated by using a degree-day factor:

$$S = K T \tag{6}$$

where

S = snowmelt in inches

K = degree-day snowmelt factor (inches/day/°C)

T = mean daily temperature, °C

A degree-day snowmelt factor K = 0.18 in/day/°C (0.10 in/day/°F) was used in all calculations. This is approximately the mid-range of the values quoted by Linsley, Kohler and Paulhus, (0.06 - 0.15in/day/°F)(*3*). The snowmelt calculated in this manner was used to estimate the antecedent moisture condition and the SCS curve number procedure was used to estimate snowmelt runoff. The SCS National Engineering Handbook does not recommend the use of curve numbers in estimating runoff from snowmelt, because there is no way to account for frozen ground. SCS considers the entire snowmelt as computed by equation (*6*) to be runoff, which is good practice when one is concerned with floods. Because this study was not concerned with floods, it was deemed more appropriate to use the curve number procedure to estimate snowmelt runoff despite its limitations. Obviously, the snowmelt runoff as calculated may have significant errors.

SIMULATION PROCEDURE

Data

The daily precipitation data and temperature data required for the simulations were obtained on magnetic tape from the National Climatic Center, Environmental Data Service, NOAA, U. S. Dept. of Commerce, at Asheville, N. C. The data set obtained is termed Day Deck 345. The normal period of record was from January 1948, through December 1973. A year beginning on March 1 was used in all simulations. The stations used are listed in Table 5.

Simulations were performed only for stations east of the Rocky Mountains for the following reasons:

1. This report is intended to cover only nonirrigated cropland and much of the cropland in the West is irrigated.
2. Rainfall gradients tend to be very steep in the West because of orographic effects. Therefore, interpolation between widely separated meteorologic stations would be misleading.

Computer Program

The program SCSRO (Soil Conservation Service Runoff) was written in FORTRAN IV. A generalized flow chart is shown in Fig. 2.

Assigning Hydrologic Soil Groups to Land Resource Areas (LRAs)

Land Resource Areas (LRAs) are shown in the map in Fig. 2, Vol. I and are discussed in Section 3.1, Vol. I. Although Land Resource Areas are defined as geographic areas characterized by a particular pattern of soil type, topography, climate, water resources, land use and type of farming (*1*) they are large enough that each of these factors varies significantly within the area. Therefore, it is impossible to characterize an entire LRA by a single soil series. In many cases, however, the major soil series listed for a LRA have similar hydrologic characteristics in that they fall into one hydrologic soil group. Where there is a wide range of hydrologic characteristics within a LRA the hydrologic soil group of the *predominant agricultural soil* was used.

The simulation results shown in Vol. I and in this Appendix should not be considered representative of the entire LRA. However, they are representative of the predominant agricultural soils of the LRA, subject, of

Table 5.—Meteorological records used in simulations

Location	Period of record	Missing years	Total years
Wichita, KA	48-67	none	20
Columbia, MO	48-67	"	20
Dodge City, KA	48-67	"	20
Kansas City, MO	43-67	"	25
Springfield, MO	43-67	"	25
Chicago, IL	43-67	"	25
Cleveland, OH	48-67	"	20
Columbus, OH	48-67	"	20
Lansing, MI	49-53, 60-69	"	15
Sault Ste. Marie, MI	47-66	"	20
Green Bay, WI	50-69	"	20
Fargo, ND	48-67	"	20
LaCrosse, WI	48-67	"	20
Des Moines, IA	46-70	"	25
Grand Island, NB	50-70	"	21
Huron, SD	43-67	"	25
Omaha, NB	48-67	"	20
Sioux Falls, SD	43-67	"	25
Bismark, ND	48-68	62	20
Williston, ND	35-62	48, 49, 50	25
Scottsbluff, NB	48-67	none	20
Rapid City, SD	49-68	"	20
Cairo, IL	30-67	48-53, 56-62	25
Indianapolis, IN	48-67	none	20
Lexington, KY	48-67	"	20
Springfield, IL	43-67	"	25
Savannah, GA	51-71	52	20
Miami, FL	48-68	49	20
Houston, TX	48-68	49	20
Brownsville, TX	48-68	49	20
Raleigh/Durham, NC	48-68	49	20
New Castle/Wilmington, DE	48-68	49	20
Charleston, SC	48-68	49	20
Columbia, SC	48-68	49	20
Jacksonville, FL	48-68	49	20
Memphis, TN	48-68	49	20
Mobile, AL	48-69	49, 62	20
Lake Charles, LA	48-58	49	10
Dallas, TX	48-68	49	20
Little Rock, AR	48-68	49	20
Oklahoma City, OK	48-67	none	20
Buffalo, NY	48-69	49, 65	20
Newark, NJ	48-68	49	20
Boston, MA	48-68	49	20
Portland, ME	48-68	49	20
Syracuse, NY	48-68	49	20
Wilkes-Barre/Scranton, PA	49-53, 56-71	50	20
El Paso, TX	48-68	49	20
Amarillo, TX	48-68	49	20
Cape Hatteras, NC	57-69	58	12
Tallahassee, FL	48-68	49	20
Pittsburgh, PA	48-68	49	20

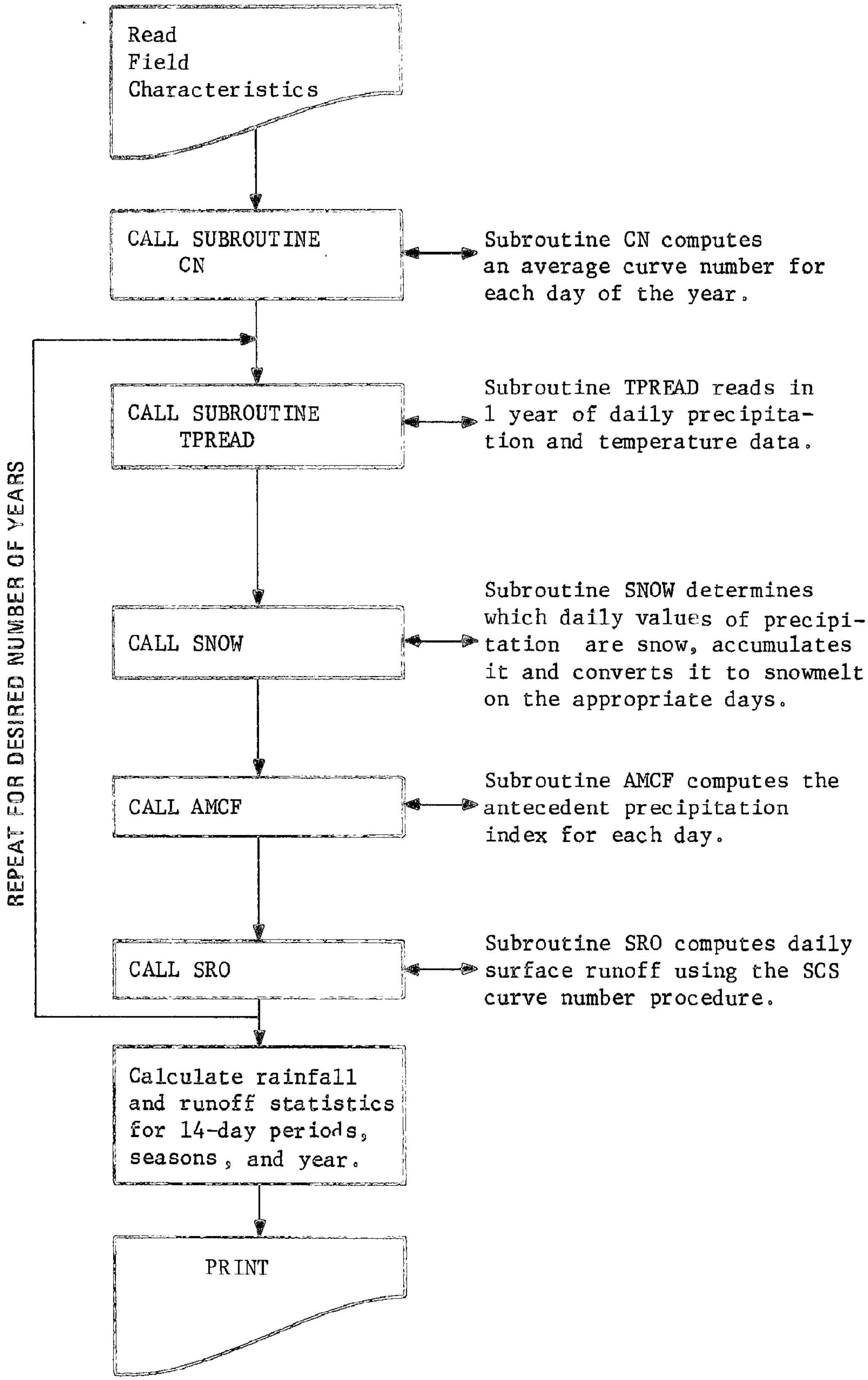

Figure 2.—Program SCSRO flowchart.

course, to the limitations of the SCS runoff estimation procedure. Predominant agricultural soil series in each LRA were obtained from Austin (*1*) and hydrologic classifications for the soil series were obtained from Table 7.1 of the SCS National Engineering Handbook (*4*). A list of the LRA's and the assigned hydrologic soil groups is presented in Table 6. These assignments were reviewed and modified by personnel of the Technical Service Centers of SCS and their assistance is gratefully acknowledged.

Table 6.–Hydrologic soil group and available water holding capacities for predominant agricultural soils in land resource areas

Land resource area	Dominant hydrologic soil group	Available water capacity in 4-ft. root zone
		(inches)
1	Mountains	
2	B	8
3	Mountains	
4	Soil Information Lacking	
5	Forest	
6	Mountains	
7	B	8
8	B	8
9	C	8
10	C	8
11	B	8
12	Mountains	
13	B	8
14	B	8
15	D	6
16	D	6
17	D	6
18	D	6
19	D	6
20	Mountains	
21	D	6
22	Mountains	
23	Soil Information Lacking	
24	" " "	
25	D	6
26	D	6
27	D	6
28	Desert	
29	"	
30	"	
31	Irrigated Desert	
32	B	8
33	Mountain	
34	B	8
35	B	8
36	C	6
37	B	8
38	Mountain	
39	"	
40	Soil Information Lacking	
41	" " "	
42	B	8
43-51	Mountains	
52	B	8
53	B	8
54	B	*6
55	B	8
56	D	4
57	B	8

Table 6.–Hydrologic soil group and available water holding capacities for predominant agricultural soils in land resource areas–Continued

Land resource area	Dominant hydrologic soil group	Available water capacity in 4-ft. root zone
		(inches)
58	B	*6
59	C	8
60	D	4
61	B	8
62	Mountains	
63	D	4
64	B	8
65	A	2
66	B	8
67	B	8
68	B	8
69	B	8
70	C	6
71	B	8
72	B	8
73	B	8
74	B	8
75	B	8
76	D	*4
77	C	8
78	C	8
79	A	4
80	C	8
81	D	6
82	C	8
83	D	6
84	B	8
85	D	6
86	D	6
87	D	6
88	Forest	
89	Forest	
90	B	8
91	A	2
92	Forest	
93	Forest	
94	Forest	
95	B	8
96	A	2
97	B	4
98	B	8
99	D	6
100	B	8
101	B	8
102	B	8
103	B	8
104	C	8
105	B	8
106	B	8

Table 6.–Hydrologic soil group and available water holding capacities for predominant agricultural soils in land resource areas–Continued

Land resource area	Dominant hydrologic soil group	Available water capacity in 4-ft. root zone
		(inches)
107	B	8
108	B	8
109	C	*6
110	C	*6
111	C	8
112	D	*4
113	D	*4
114	D	4
115	B	8
116	C	*4
117	Mountains	
118	D	6
119	Mountains	
120	C	*4
121	C	8
122	B	8
123	C	8
124	C	8
125	Mountains	
126	C	8
127	Mountains	
128	B	8
129	B	8
130	Mountains	
131	D	6
132	D	6
133	B	8
134	C	8
135	D	6
136	B	8
137	A	4
138	B	4
139	C	*4
140	C	8
141	C	8
142	D	6
143	Mountains	
144	A	4
145	B	8
146	C	8
147	B	8
148	C	8
149	C	8
150	D	6
151	Swamp	
152	D	6
153	C	6
154	A	4
155	B	8
156	Swamp	

*Available water-holding capacity reduced because root zone is shallower than 4 feet.

SIMULATION RESULTS

Program SCSRO output for each 14-day period for the n years of record may include:

1. A listing of rainfall amounts ordered by magnitude.
2. A listing of simulated runoff ordered by magnitude.
3. The mean and standard deviation of rainfall and runoff events.

The statistical summary for the n-year simulation included:

1. A table showing the number of runoff events for each 14-day period for each year.
2. The probability that there would be no runoff events in any year for each 14-day period.
3. The mean annual simulated runoff.
4. The mean growing season simulated runoff.

Maps of the mean annual and seasonal simulated runoff (potential direct runoff) are shown for each hydrologic soil group in Figs. 3 through 10. Because of the relatively small area of soils classified in hydrologic group A, simulations with this group were not performed for all rainfall stations. The growing season was taken as the time interval between emergence and harvest and varied with location. These maps can be used to supplement the information presented in Figs. 3 and 4 in Vol. I.

Too few rainfall stations were used in this analysis to depict climatic and orographic influences in the Appalachian Mountains; therefore, care must be used in interpreting the maps in these regions.

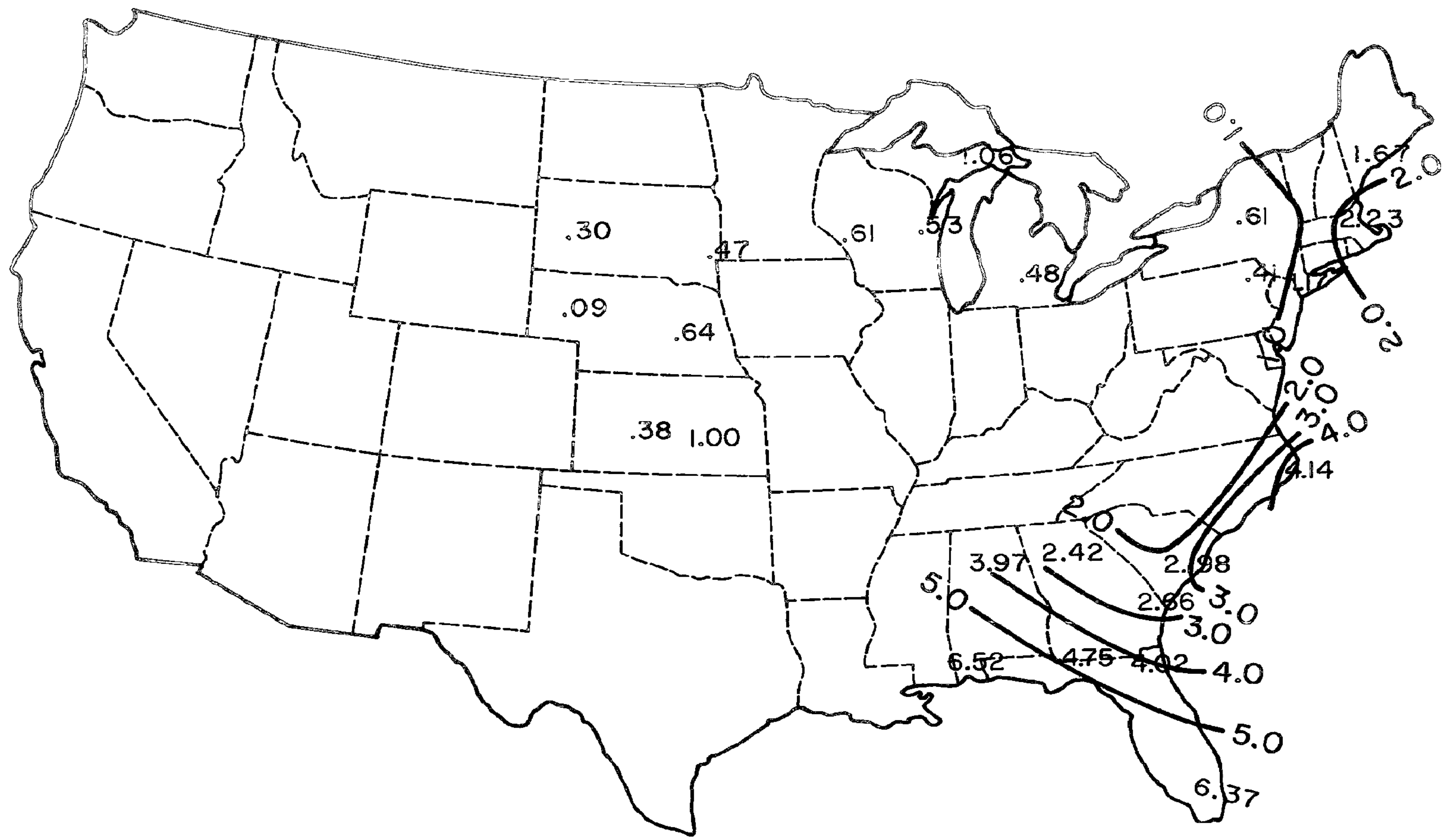

Figure 3.–Mean annual potential direct runoff in inches. Straight-row corn in good hydrologic condition–Hydrologic Soil Group A.

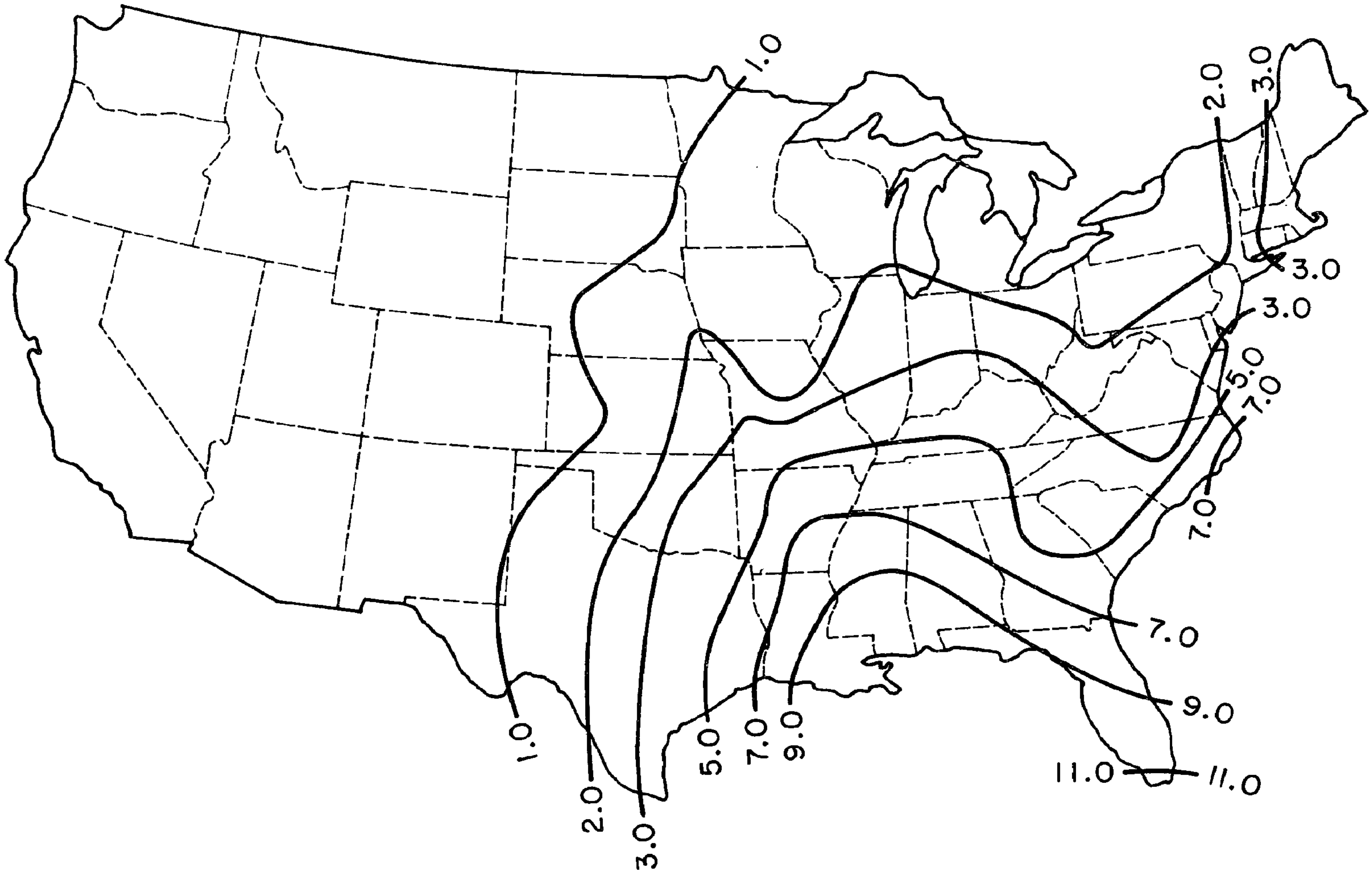

Figure 4.–Mean annual potential direct runoff in inches. Straight-row corn in good hydrologic condition–Hydrologic Soil Group B.

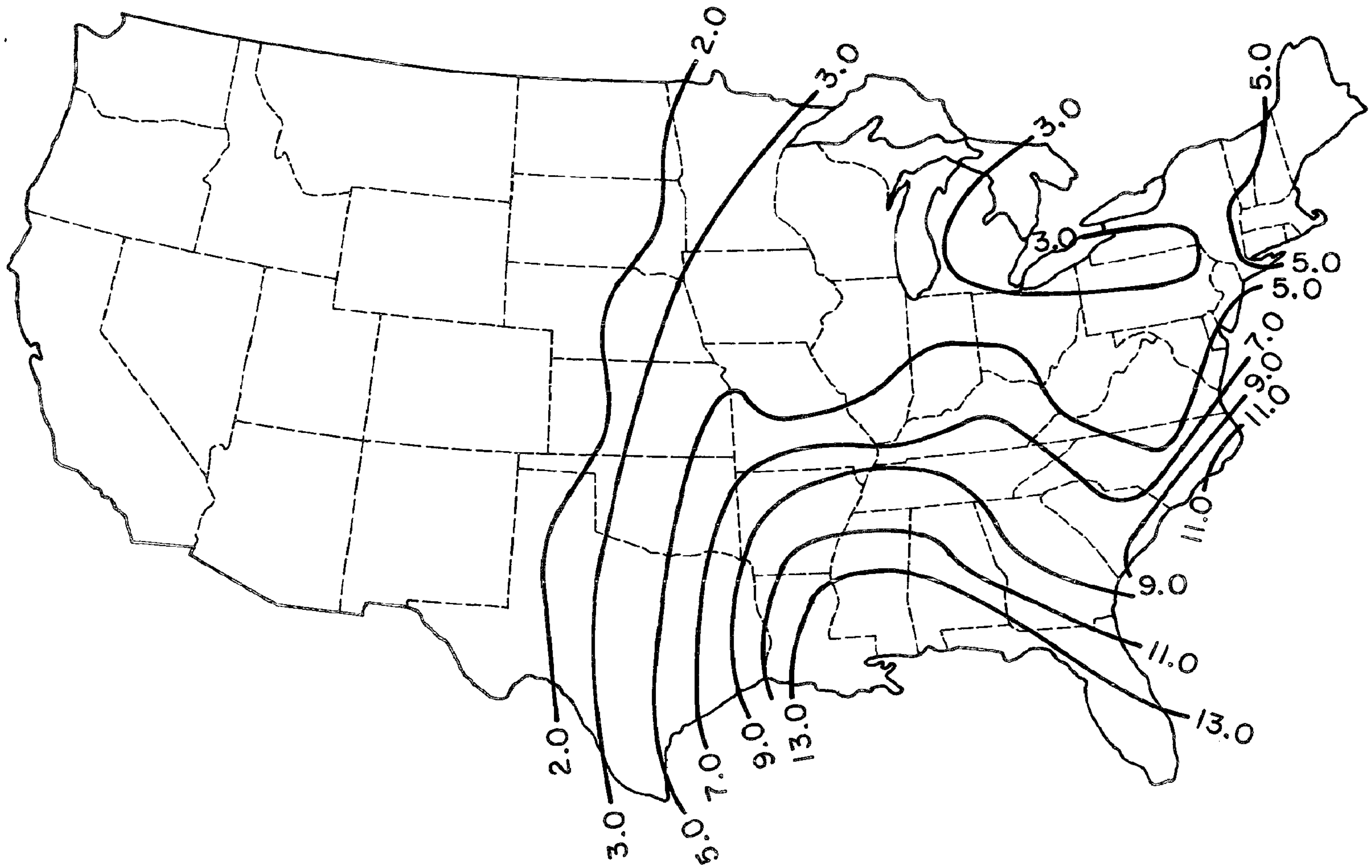

Figure 5.–Mean annual potential direct runoff in inches. Straight-row corn in good hydrologic condition–Hydrologic Soil Group C.

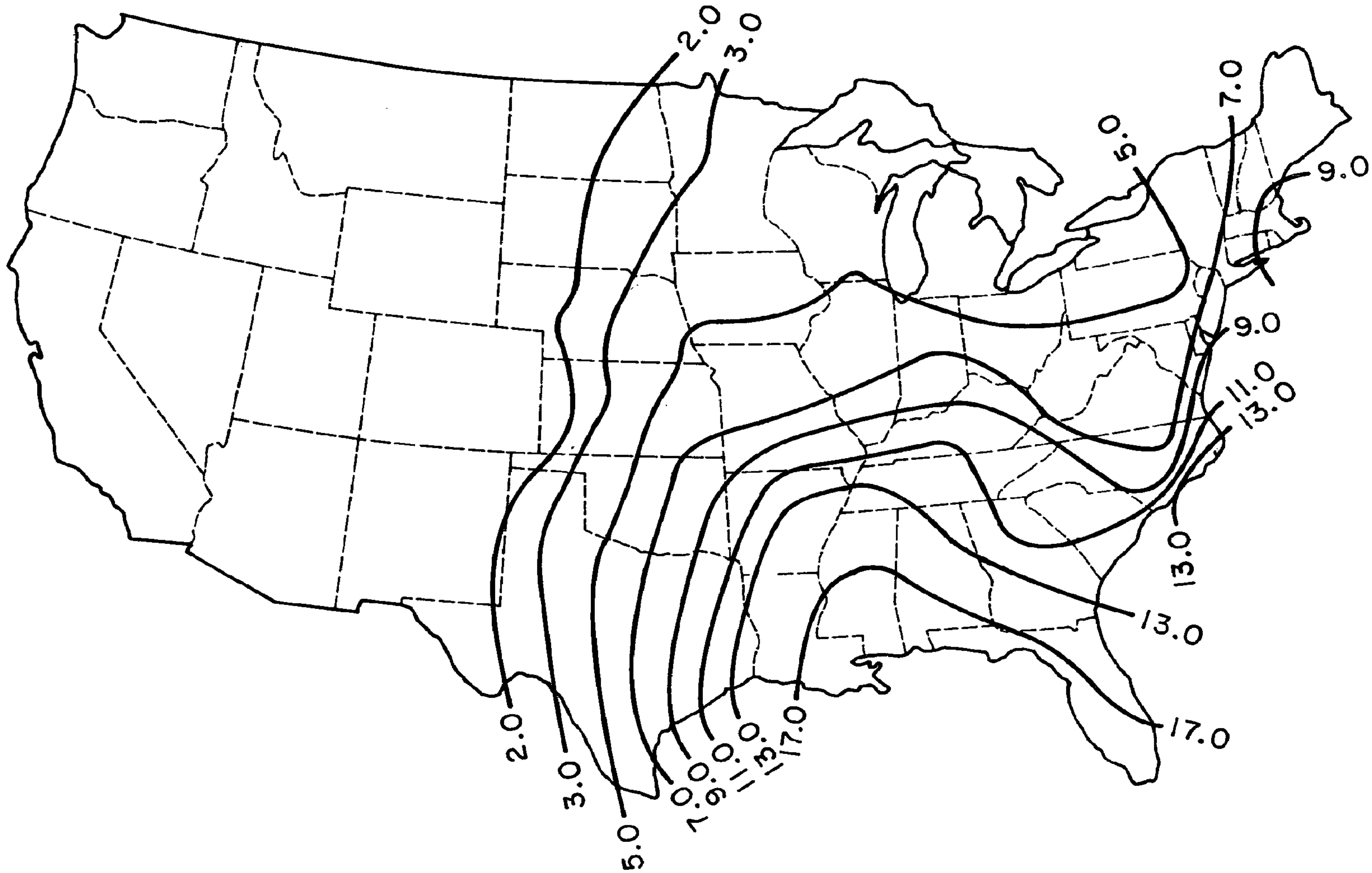

Figure 6.—Mean annual potential direct runoff in inches. Straight-row corn in good hydrologic condition—Hydrologic Soil Group D.

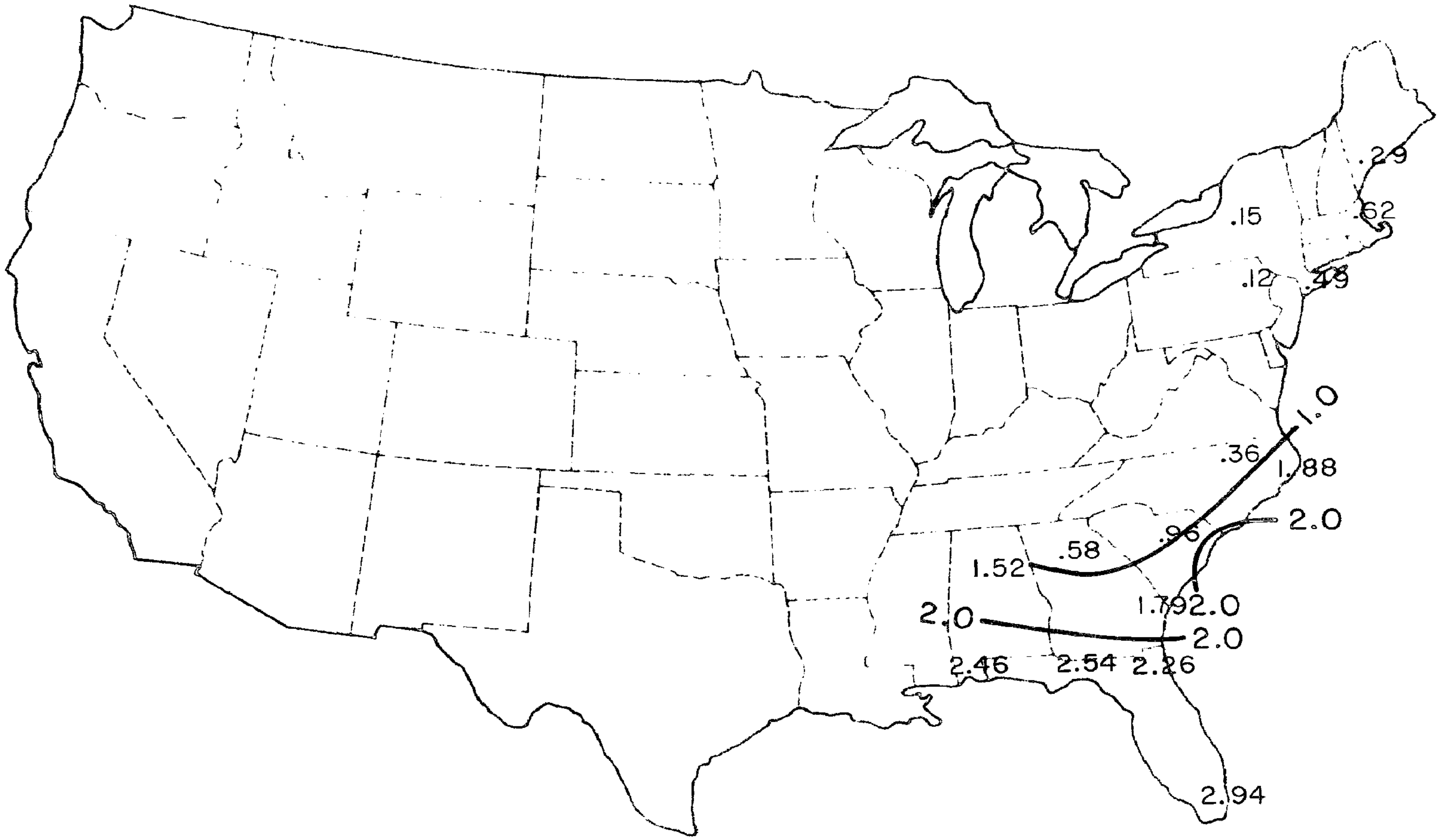

Figure 7.—Mean growing season potential direct runoff in inches. Straight-row corn in good hydrologic condition—Hydrologic Soil Group A.

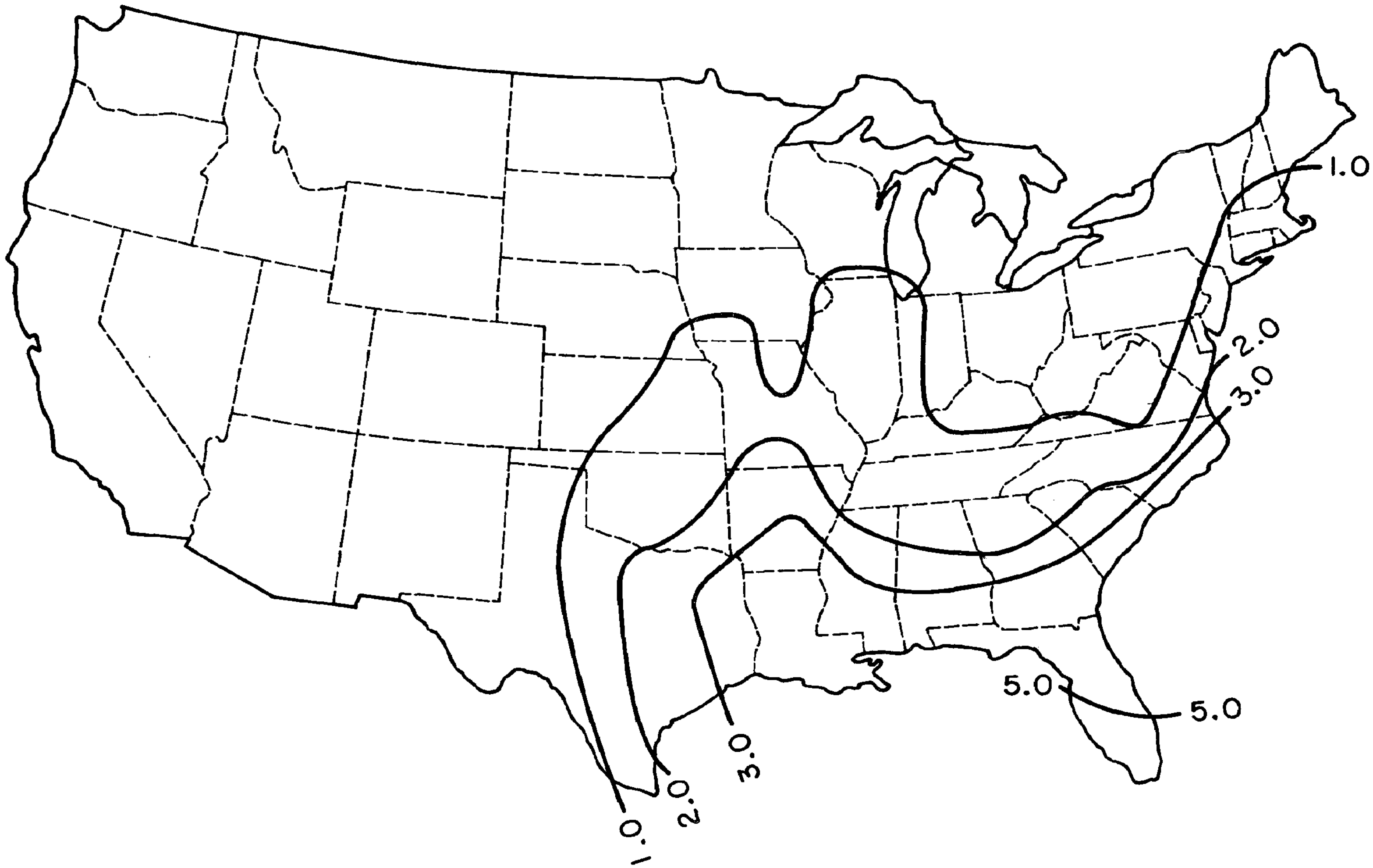

Figure 8.–Mean growing season potential direct runoff in inches. Straight-row corn in good hydrologic condition–Hydrologic Soil Group B.

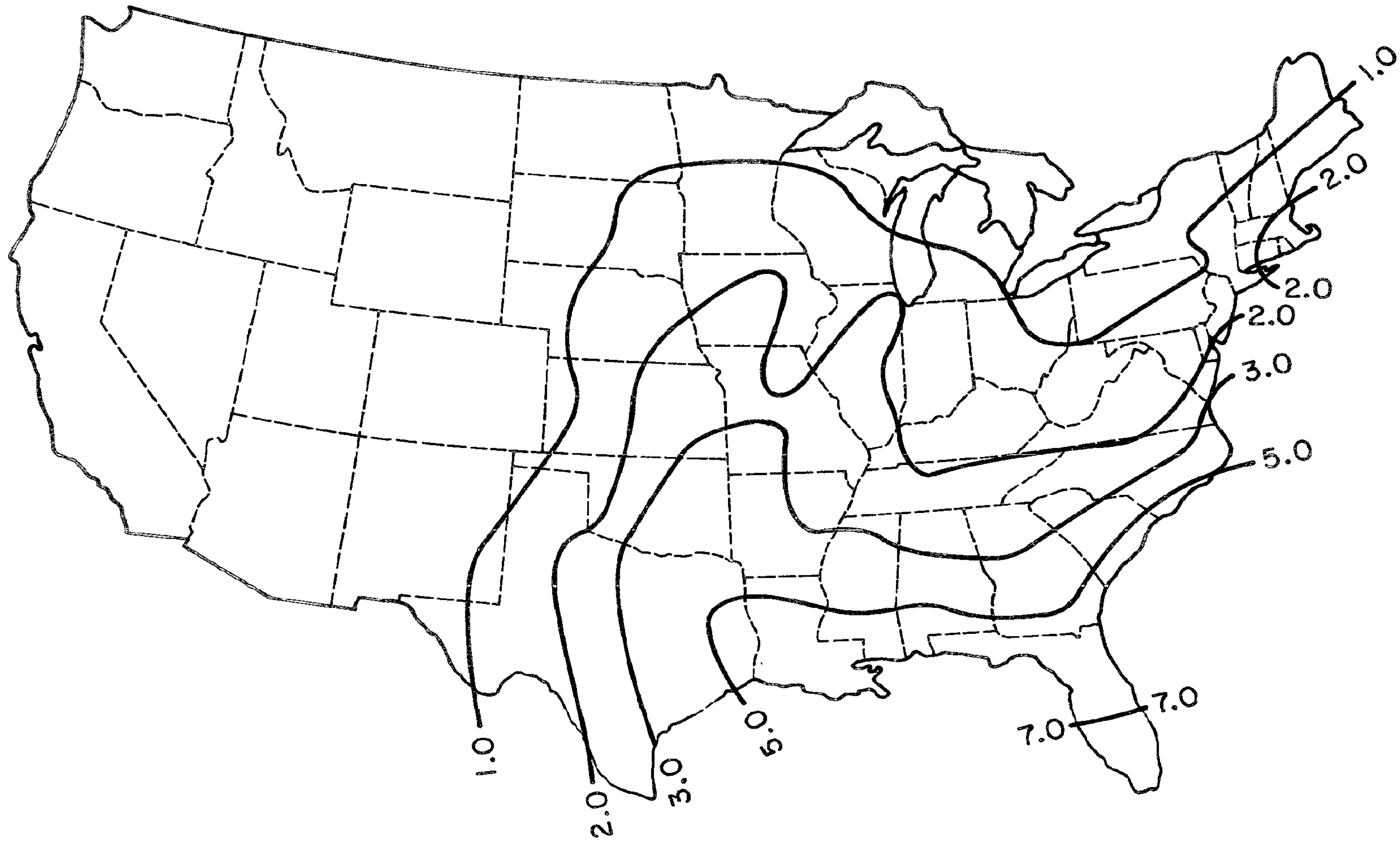

Figure 9.–Mean growing season potential direct runoff in inches. Straight-row corn in good hydrologic condition–Hydrologic Soil Group C.

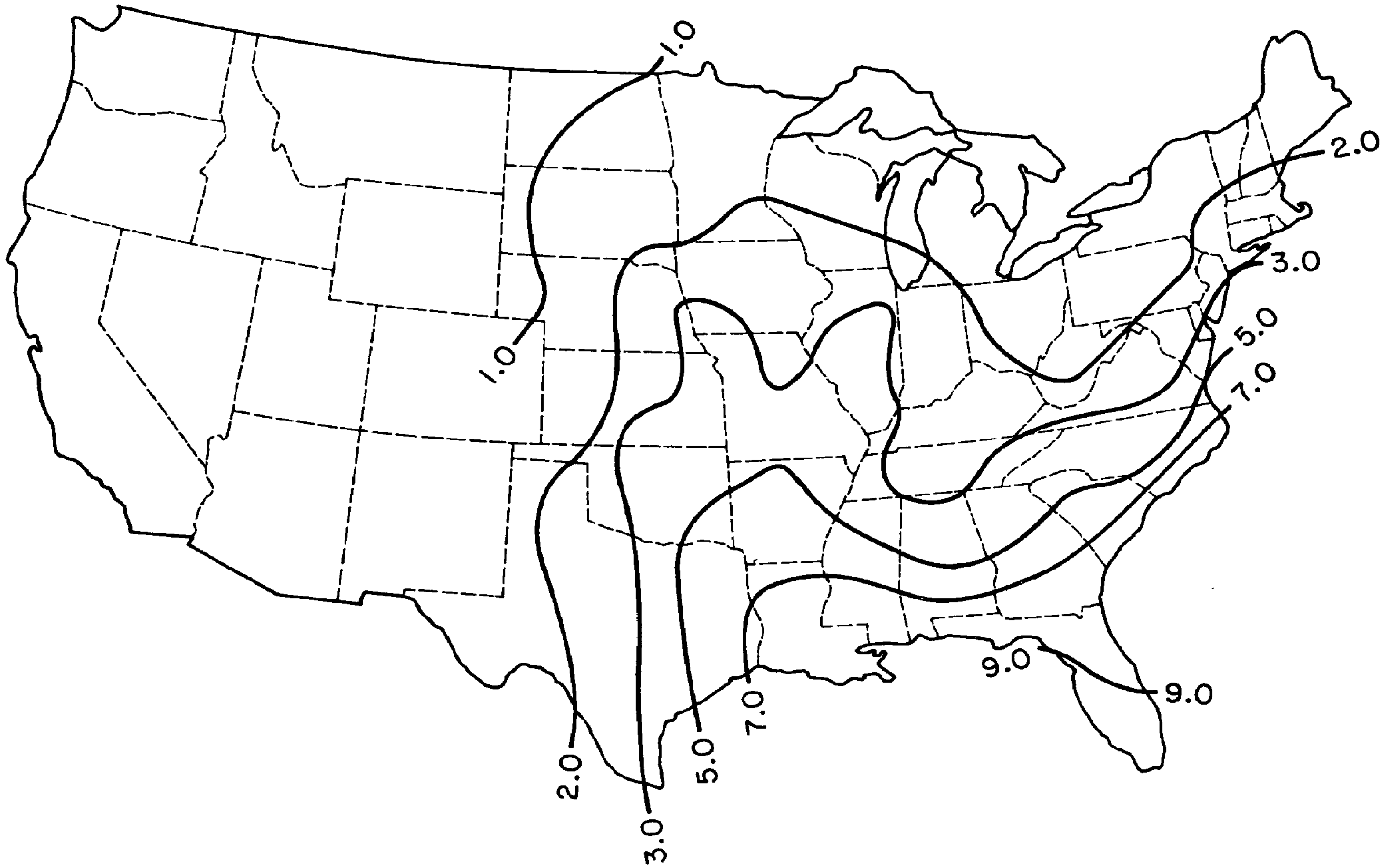

Figure 10.–Mean growing season potential direct runoff in inches. Straight-row corn in good hydrologic condition–Hydrologic Soil Group D.

CONSISTENCY CHECK AND DISCUSSION OF ERRORS

The accuracy of the simulated runoff amounts was checked by comparing average annual potential direct runoff with measured average annual runoff from several small, single-crop watersheds in straight-row crops (primarily corn and cotton) (*7, 8*). The watersheds used in this comparison and a brief data summary are presented in Table 7. The following linear regression equation was obtained:

$$Q_s = 1.365 + 0.578\, Q_o \qquad (7)$$

where

Q_s = simulated average annual direct runoff.

Q_o = observed average annual runoff.

The coefficient of variation (r^2) was 0.616. A scatter diagram of computed versus observed average annual runoff is shown in Fig. 11. The vertical lines emanating from the plotted points indicate the range in simulated runoff that would occur if the soils were assumed to belong to adjacent hydrologic soil groups. This indicates

Table 7.–Surface runoff consistency check

Watershed		Area	Crop	Hydrologic soil group	Mean annual runoff (in.) Observed	Simulated
		acres				
College Park, MD	W-3	6.06	Soybeans	B	2.47	2.60
Americus, GA	W-1	17.9	Sweet corn Corn	B	1.27	7.17
Lafayette, IN	W-4	2.01	Cotton Corn	B	2.26	2.60
" "	W-5	2.87	Soybeans Corn	84% B 16% C	4.14	3.05
" "	W-8	1.96	Soybeans Corn	B	4.89	2.60
" "	W-10	2.06	Soybeans Corn	66% B 34% C	5.88	3.57
" "	W-12	3.37	Soybeans Corn	89% B 11% C	3.98	2.91
" "	W-13	3.02	Soybeans Corn	80% B 20% C	3.93	3.17
" "	W-15	3.59	Soybeans Corn	B	4.33	2.60
Clarinda, IA	W-V	3.25	Soybeans Corn	90% B 10% D	1.41	2.32
" "	W-W	1.97	Corn	90% B 10% D	3.37	2.32
" "	W-Y	3.25	Corn	72% B 28% D	1.06	2.89
Coshocton, OH	W-115	1.61	Corn	C	2.85	3.05
" "	W-110	1.27	Corn	C	2.41	3.05
" "	W-118	1.96	Corn	C	1.97	3.05
" "	W-192	7.59	Corn	C	3.13	3.05
" "	W-106	1.56	Corn	C	3.23	3.05
Guthrie, OK	W-2	3.21	Cotton	B	7.67	2.30
Garland TX	W-III	10.4	Cotton	D	11.67	8.30
Spur, TX	W-2	9.39	Corn Cotton	50% B 50% C	2.70	2.05
Riesel, TX	Y-7	40.0	Row crops	D	7.00	8.75
Hastings, NB	3-H	3.95	Corn	B	4.85	1.65
Oxford, MS	WC-1	3.88	Corn	C	14.99	11.00
" "	WC-3	1.61	Corn	C	13.77	11.00
Chickasha, OK	C-1	17.8	Corn	81% C 19% B	1.24	3.32

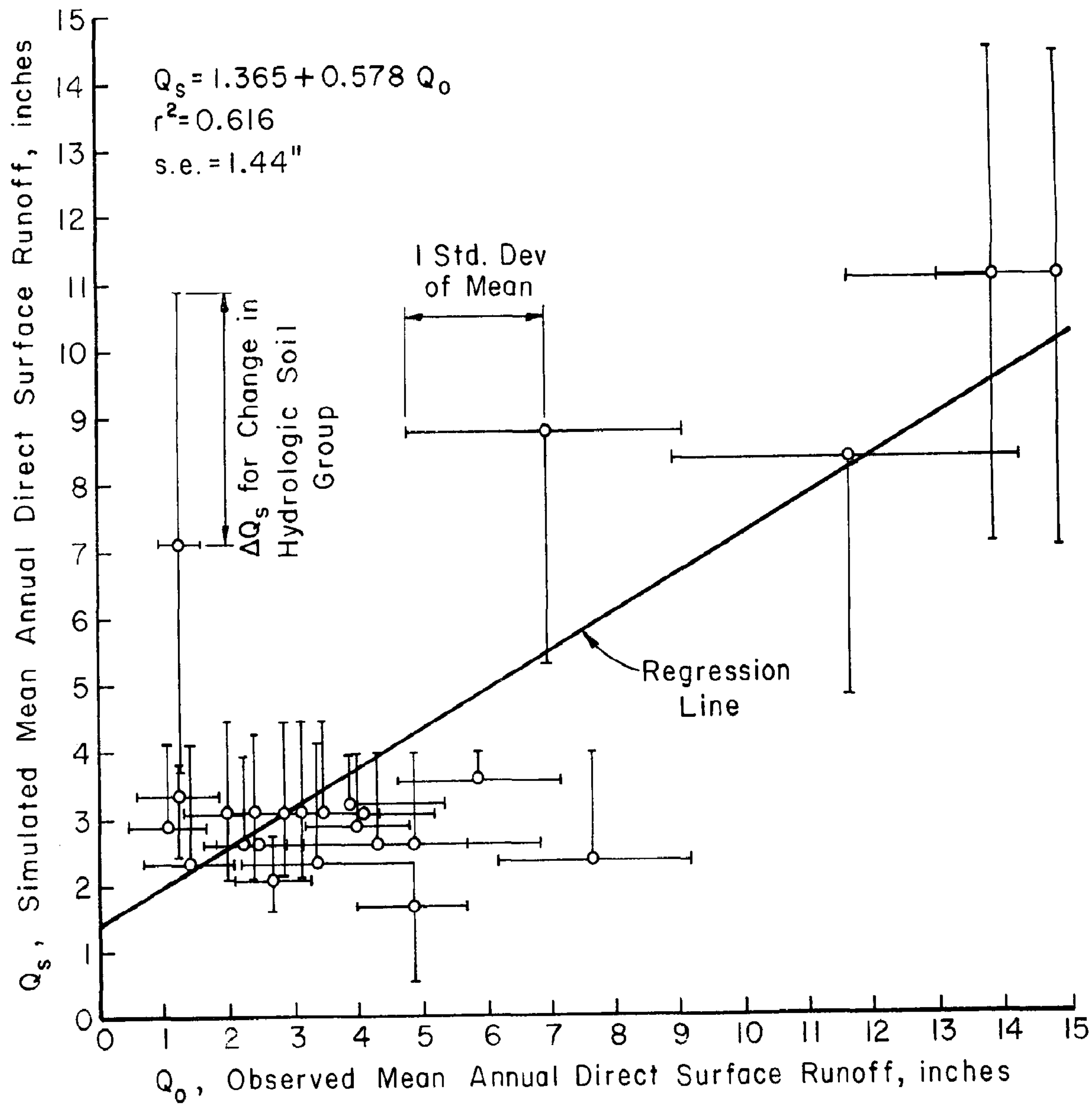

Figure 11.–Simulated and observed mean annual direct surface runoff.

the inherent limitation in the SCS method caused by lumping all soils into four distinct groups. Horizontal bars emanating from plotted points indicate estimated standard errors of the mean of observed data. This illustrates the problem of short, fragmented records. The SCS method tended to underestimate runoff of more than 3 inches.

Meaningful comparisons are difficult because the observations were made over such a long time. Agricultural practices have changed drastically so the data are not stationary. For example, hybrid corn and higher fertility levels have led to rapid canopy establishment and more residues after harvest. The simulated condition after harvest—approximately 67% cover by residues—probably is not consistent with the practices on watersheds where the data were obtained. One would anticipate that the simulated runoff would be less than the observed in this case. Data from some of the watersheds listed in Table 7 were undoubtedly used in developing the SCS curve number procedure so this is not an independent test of its predictive capabilities.

Although the relationship between simulated and observed direct runoff shown in Fig. 11 is not as good as one would wish, it must be compared with the available alternatives before one can judge its usefulness. One alternate that has been suggested is to use the map of surface-water runoff prepared by the U.S. Geological Survey (*6*) as an indicator of potential loss by direct runoff. To test this method, consider the following regression relationship between the average annual runoff from the USGS map for the locations in Table 7 and the observed average annual direct runoff:

$$Q_G = 6.74 + 0.503\, Q_o\,; \quad r^2 = 0.138 \qquad (8)$$

where Q_G is average annual surface-water runoff from the USGS map.

The simulated results obviously are superior to those obtained from the runoff map as indicators of potential direct runoff.

One would anticipate that the sum of the simulated average annual direct runoff and the average annual deep percolation estimated by the procedures described in Appendix B of this volume should be rather well correlated with the average annual streamflow from the USGS maps.

The following regression equation was obtained between simulated direct runoff plus percolation and runoff (streamflow) from the USGS map for 45 of the 52 meteorological stations used:

$$(Q_s + Q_p) = 0.409 + 0.979\, Q_G\,; \quad r^2 = 0.884 \qquad (9)$$

where Q_p is the simulated average annual deep percolation. Seven stations in the karst area of Florida and in the coastal area of the Southeastern United States were omitted because anomalies on the USGS map indicate that much of the groundwater runoff flows directly into the ocean. These crude checks indicate that the simulations provide reasonable estimates of annual direct runoff and percolation.

Records of runoff from continuous straight-row corn were not readily available for periods shorter than one month so it was not possible to check the accuracy of the time distribution of simulated potential direct runoff within the year. However, 20 years of runoff data were available for a small watershed in meadow at Coshocton, Ohio. Direct runoff for 14-day periods simulated by the SCS procedure was compared with observed data. Simulated and observed mean runoff per event, mean number of runoff events per period, and mean runoff amount per period are shown in Fig. 12. The standard deviation of the observed runoff per period is indicated by a vertical line for each period.

The simulated runoff per period is within one standard deviation of the observed runoff for 14 of the 26 periods. Assuming that the mean value is normally distributed for each period and that the 26 periods are independent trials, the null hypothesis cannot be rejected at the 10 percent level.

Sample distribution functions of runoff amount per event for two periods are shown in Fig. 13. A Kolmogorov-Smirnov test comparing the distribution functions indicates that the null hypothesis cannot be rejected at the 10 percent level for both periods.

Although it is impossible to make strong inferences on the basis of the limited tests performed, the SCS method appears adequate for arriving at a first estimation of direct surface runoff.

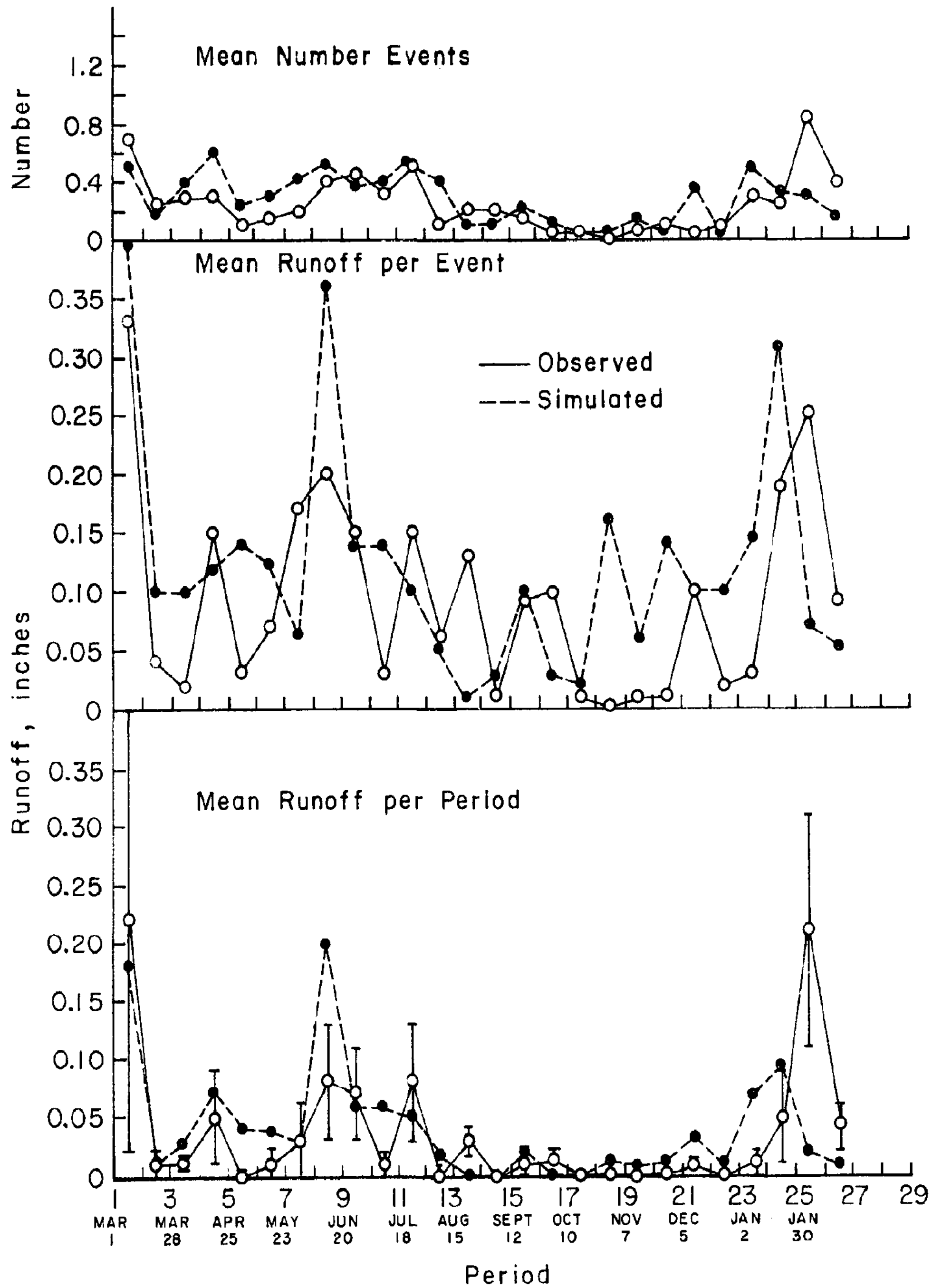

Figure 12.–Comparison of simulated and observed runoff records by 14-day periods. Coshocton, Ohio meadow.

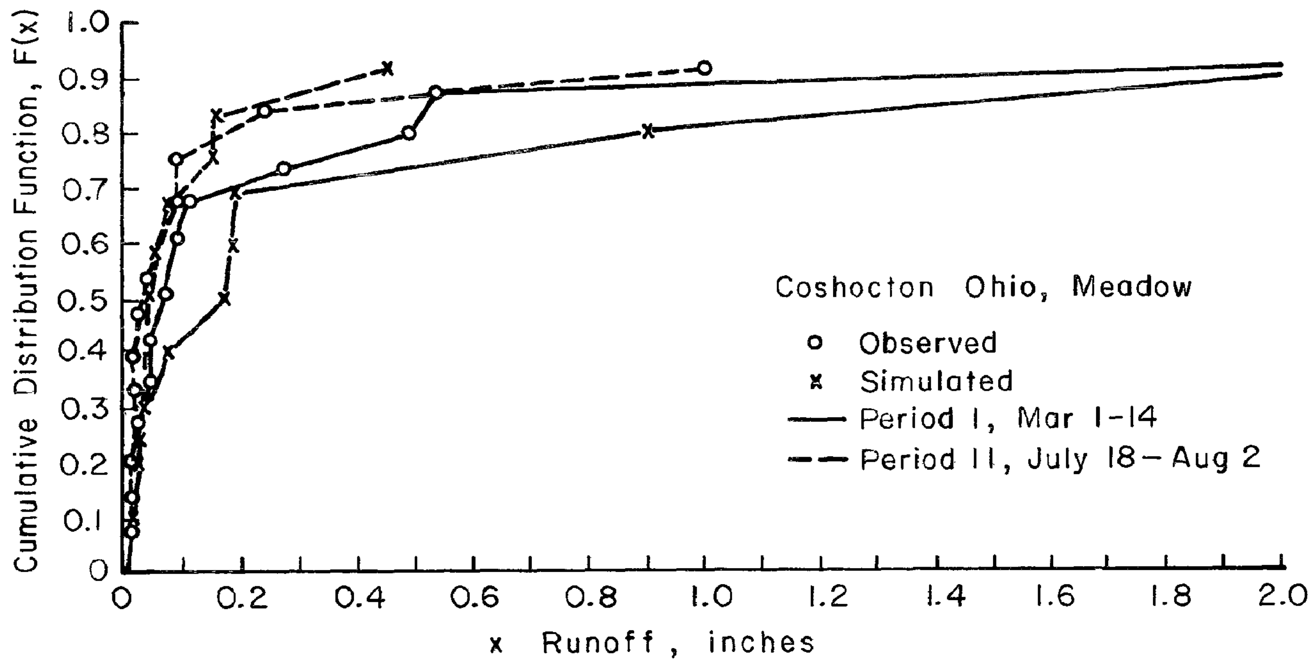

Figure 13.—Sample distribution functions for simulated and observed records. Coshocton, Ohio meadow.

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APPENDIX B

SIMULATION OF POTENTIAL PERCOLATION AND NITRATE LEACHING

INTRODUCTION

Potential percolation was defined in Section 3.4, Volume I, as the annual amount of water that would percolate below the root zone in a field of straight-row corn.

The relationship between potential percolation and other hydrologic variables can be expressed by the vertical water balance equation for a column of soil as

$$P = Q_s + E + Q_p + \Delta S \qquad (1)$$

where P is the precipitation, Q_s is the direct runoff, E is the evapotranspiration, Q_p is the percolation from the bottom of the root zone and ΔS is the change in soil water storage in the root zone during the time under consideration. In Eq. (*1*), it is assumed that imported water, lateral porous media flow, and change in surface detention storage are negligible.

Soluble agricultural chemicals that are not strongly adsorbed by the soil may be carried below the root zone by percolating water. After the percolating water has reached the ground water table, it will move laterally and eventually reappear in a stream, lake or possibly the ocean. Some ground water may also flow into the root zone in seepage areas and be transpired by vegetation.

Because precipitation can be considered as a stochastic process, all of the other variables in Eq. (*1*) are stochastic in nature. A simulation approach is necessary to estimate each of the other terms because of the rather complex relationships between them. To carry out the simulation we need a soil water model, a direct runoff model and an evapotranspiration model. The SCS procedure as described in Appendix A was used as the direct runoff model. The soil-water model and the evapotranspiration model are described in this Appendix.

THE SOIL-WATER MODEL

The soil-water model described in this section utilizes the approximation that soil-water moves readily under gravitational forces when its water content is above field capacity. It is further assumed that water does not move downward when the water content is below field capacity and that when the water content reaches the wilting point it is no longer available to plants.

The structure of the three-compartment soil-water model used is shown in Fig. 1. The water content of compartment i is designated as $S_i(t)$ and the maximum capacity of the i^{th} compartment is K_i, which corresponds to field capacity. Compartments 1 and 2 represent the active root zone and compartment 3 represents the water storage below the current root zone and above the maximum depth of rooting, d_3. The depth of the surface layer is d_1 and $d_2(t)$ is the time varying depth of the root zone. The extraction of water from different depth zones varies with stage of crop development, so the capacities of compartments 2 and 3 vary with time as the crop canopy expands but their sum is a constant. The maximum capacities of the compartments can be interpreted as the available water-holding capacity per unit area for the depths d_1, $d_2(t)$-d_1 and d_3-$d_2(t)$.

Input to the system is $X_1(t)$, which is the difference between daily rainfall plus snowmelt and direct runoff. System output is $Y_1(t)$, $Y_2(t)$ and $Y_3(t)$ where

Y_1 (t) = daily evaporation from the soil surface
Y_2 (t) = daily transpiration
Y_3 (t) = daily seepage below the maximum root zone

The evaporation and transpiration model will be described in the next section. The rules for movement of water between compartments are as follows:

$$\left.\begin{aligned} &S_1(t+1) = S_1(t) + X_1(t+1) - Y_1(t+1) - Y_4(t+1) - y_{12}(t+1) \\ &y_{12}(t+1) = 0 \text{ if } S_1(t) + X_1(t+1) - Y_1(t+1) - Y_4(t+1) \le K_1 \\ &y_{12}(t+1) = S_1(t) + X_1(t+1) - Y_1(t+1) - Y_4(t+1) - K_1\,;\text{ otherwise} \end{aligned}\right\} \quad (2)$$

where $y_{12}(t+1)$ is the flow from compartment 1 to compartment 2.

Equation (2) states that the water content of compartment 1 in period t+1 is equal to the water content in period t plus the infiltration on day t+1 less evaporation, transpiration and flow to compartment 2. Flow from compartment 1 to compartment 2 exists only if the available waterholding capacity of compartment 1 is exceeded. No provision is made for upward flow in this model, although such flow is physically possible.

$$\left.\begin{aligned} &S_2(t+1) = S_2(t) + y_{12}(t+1) - Y_5(t+1) - y_{23}(t+1) \\ &y_{23}(t+1) = 0 \text{ if } S_2(t) + y_{12}(t+1) - Y_5(t+1) \le K_2 \\ &y_{23}(t+1) = S_2(t) + y_{12}(t+1) - Y_5(t+1) - K_2\,;\text{ otherwise.} \end{aligned}\right\} \quad (3)$$

$$\left.\begin{aligned} &S_3(t+1) = S_3(t) + y_{23}(t+1) - Y_3(t) \\ &Y_3(t) = 0 \text{ if } S_3(t) + y_{23}(t+1) \le K_3 \\ &Y_3(t) = S_3(t) + y_{23}(t+1) - K_3\,;\text{ otherwise.} \end{aligned}\right\} \quad (4)$$

Flow relationships for compartments 2 and 3 are similar to those for compartment 1 except that there is no evaporative loss from compartments 2 and 3 and no transpiration loss from compartment 3.

A schematic drawing of the seasonal variation in the root zone is shown in Fig. 2. The depth of the soil layer from which evaporation can occur, d_1, is assumed to be constant throughout the year. The total depth of the zone from which evaporation and transpiration losses occur remains at d_1 throughout the dormant period, then is approximated by a linear expansion beginning at the plant emergence date t_1, and reaches its full extension to d_3 when the crop canopy factor has reached its maximum at time t_2. The zone depth then remains constant until harvest, t_3, when it instantaneously goes back to d_1. The assumption is made for this simulation that there is no weed growth or winter cover crop withdrawing moisture from harvest until spring plowing date. While the root zone is expanding, the capacities and contents of compartments 2 and 3, $K_2(t)$, $K_3(t)$, $S_2(t)$ and $S_3(t)$ are changing each day according to the following equations:

$$K_2(t+1) = K_2(t) + \left(\frac{d_3-d_1}{t_2-t_1}\right)\theta \quad (5)$$

where θ is the volumetric available water holding capacity defined as the difference between field capacity and wilting point, and t_1 and t_2 are the emergence date and full canopy date, respectively.

$$K_3(t+1) = K_3(t) - K_2(t+1) + K_2(t) \quad (6)$$

$$S_2(t+1) = S_2(t) + \frac{[K_2(t+1)-K_2(t)]}{K_3(t)}\,S_3(t) \quad (7)$$

$$S_3(t+1) = S_3(t) - S_2(t+1) + S_2(t) \quad (8)$$

The capacity of compartment 2 increases in proportion to the increased rooting depth and the water content is increased by the amount of water in the incremental depth. The capacity and water content of the third compartment are decreased by equivalent amounts.

The soil-water model is similar in many respects to some of the simple models that have been used in irrigation and hydrology for many years (*11, 13, 27*).

This model obviously preserves mass continuity as expressed by Eq. (*1*). It does not incorporate relationships between potential gradients, hydraulic conductivity and water flux in porous media, using instead the

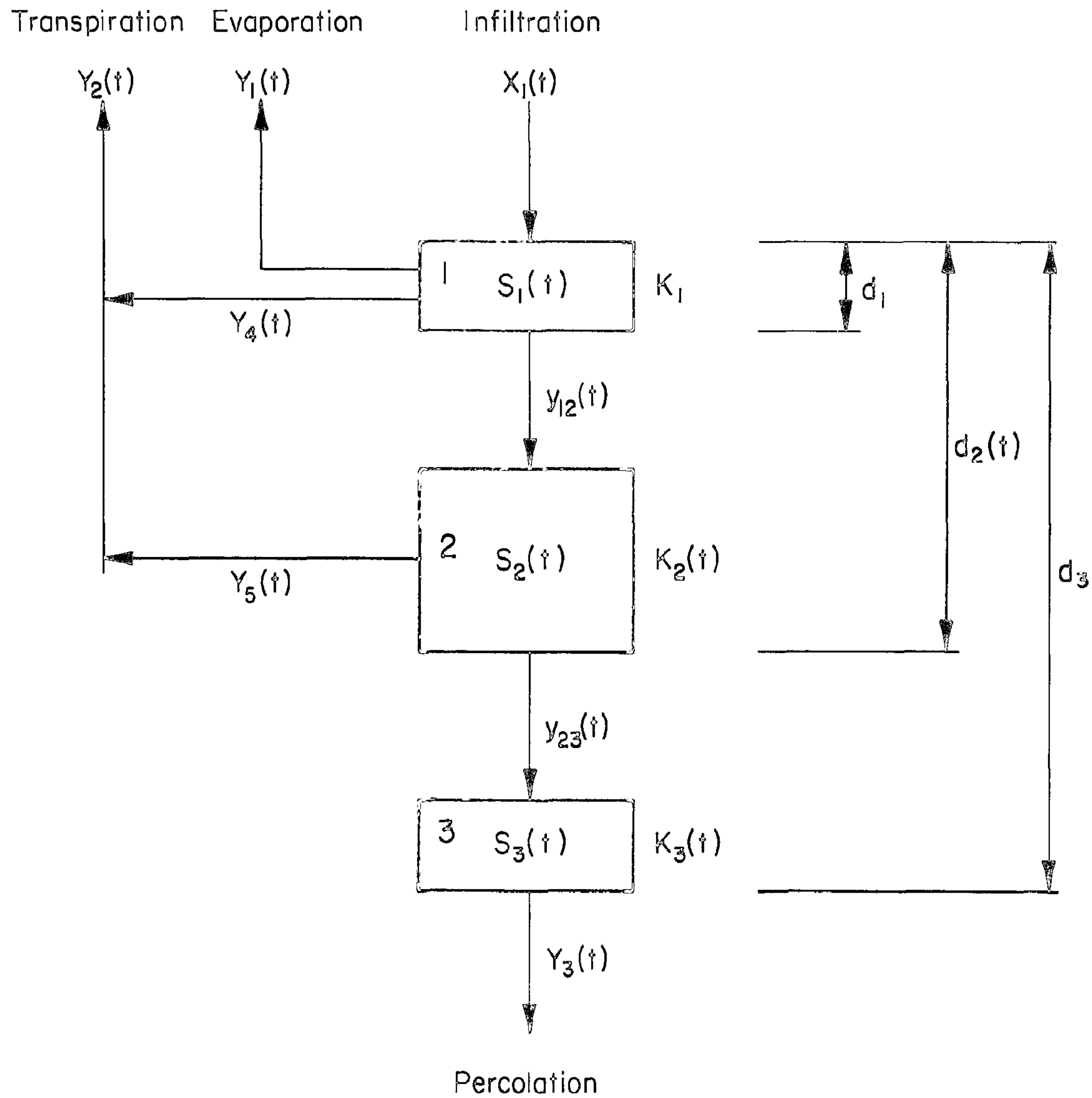

Figure 1.–Soil-water model schematic.

crude concept of "field capacity." The approximations and assumptions included in the model probably would not lead to serious errors in deep, well-drained soils. Substantial errors may occur where water tables are shallow or in soils with shallow, relatively impermeable layers.

THE EVAPOTRANSPIRATION MODEL

The evapotranspiration (ET) model is based on the frequently used assumption that ET will take place at the "potential" rate if the soil has adequate water and a complete crop canopy or if the surface is wet. Actual evapotranspiration rates will be less than potential as the soil dries.

The model can be described by the following equations:

Evaporation:

$$Y_1(t) = [1-C(t)]\ K_p\ P(t)\ \frac{S_1(t)}{K_1} \tag{9}$$

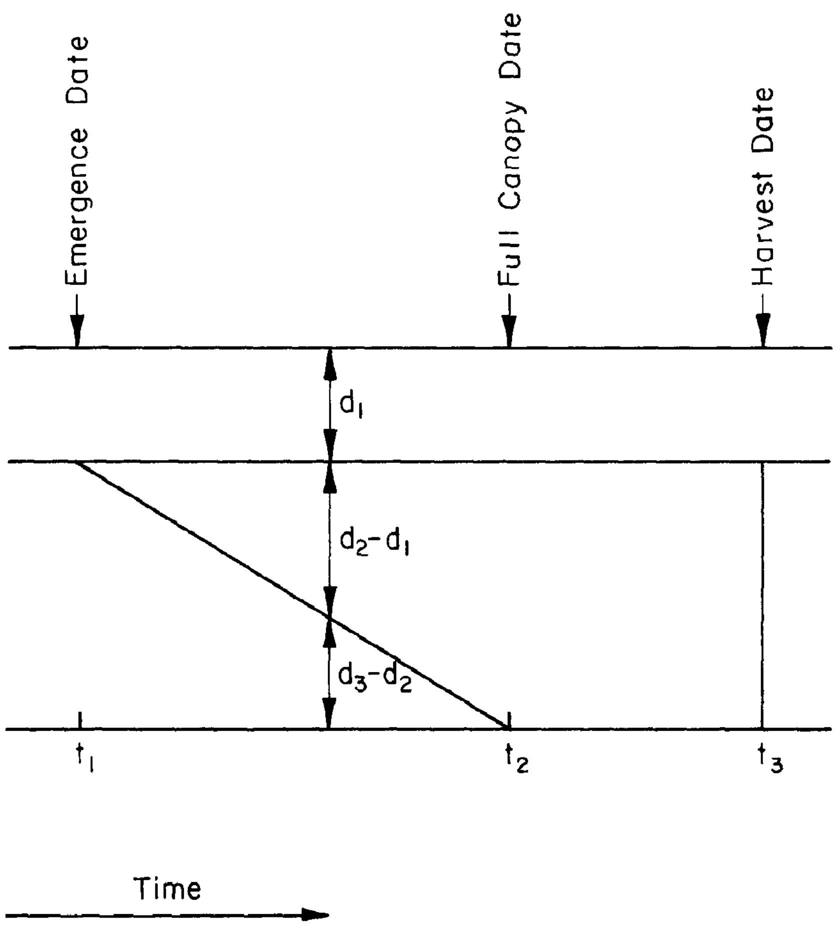

Figure 2.–Seasonal variation of root zone.

where C(t) is a time varying crop coefficient related to the portion of the soil surface covered by vegetation, K_p is a coefficient to convert pan evaporation to potential ET, P(t) is the pan evaporation in inches per day and $S_1(t)$ and K_1 have been previously defined.

Total transpiration is given by:

$$Y_2(t) = C(t)\, K_p\, P(t)\, f \ . \tag{10}$$

where f is the ratio of actual to potential evapotranspiration and depends on the water content of compartments 1 and 2 as shown in Fig. 3.

The transpiration loss from compartment 1 is:

$$Y_4(t) = Y_2(t) \frac{S_1(t)}{[K_1 + K_2(t+1)]} \tag{11}$$

and the transpiration loss from compartment 2 is:

$$Y_5(t) = Y_2(t) - Y_4(t) \ . \tag{12}$$

An examination of Eqs. (9) through (12) reveals that for bare fallow conditions, [C (t) = 0], only evaporation occurs, i.e., $Y_2(t) = 0$. For full canopy conditions $C(t) = 1, Y_1(t) = 0$ and all loss is through transpiration. The form of this evapotranspiration model is identical to that used by Hanson (*5*).

NITRATE LEACHING MODEL

The purpose of this model is to gain a quantitative insight into what percentage of nitrogen applied as ammonium in the fall or spring would move below the root zone before the roots had reached their full extension in the following crop year. Any nitrogen present in the profile at the time of fertilization is ignored as are denitrification losses. It is also assumed that there is no nutrient uptake by weeds or winter

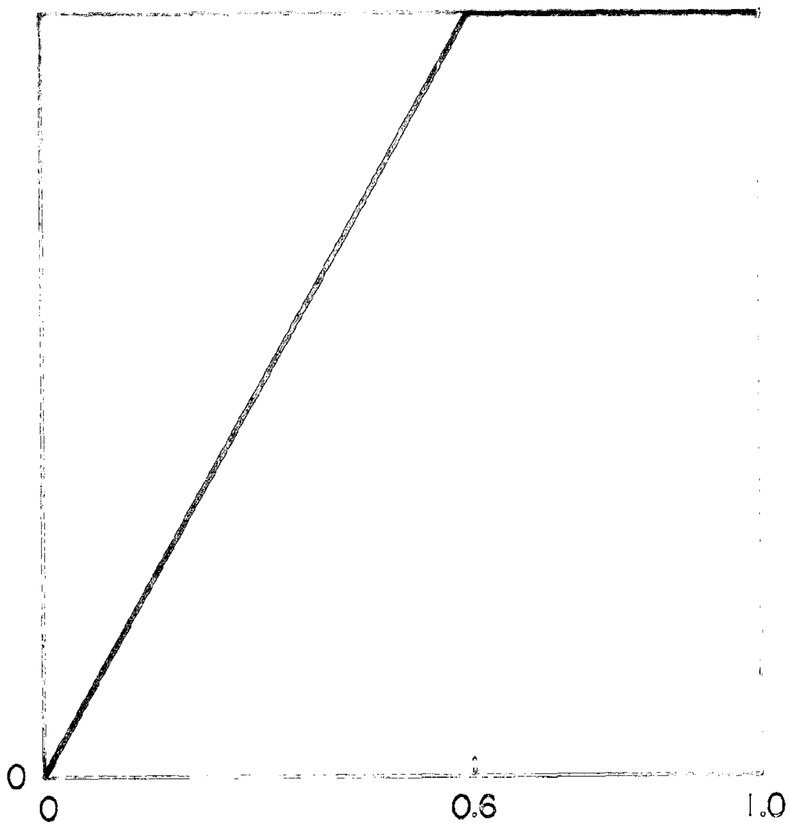

Figure 3.—Relationship between actual and potential evapotranspiration and root zone water content.

cover crops. Water flow through the soil during the dormant period is assumed to be piston flow. The upper compartment with capacity K_1 was retained in the soil model. At the time ammonium was applied in the fall, the water in compartments 2 and 3 was assumed to be uniformly distributed in the depth $d_3 - d_1$. Any time the capacity of compartment 1 is exceeded, the volume of water $y_{12}(t)$ moves as a piston flow, displacing some of the soil water ahead of it as shown in Fig. 4.

The depth increment of the i^{th} output from compartment 1 to the lower zones is given by the expression

$$Z_i = y_{12}(i)/(\theta_F - E) \tag{13}$$

where E is an exclusion factor to account for the fraction of the soil water not containing nitrate and θ_F is the volumetric field capacity (*21, 22*). E as used in these calculations is the ratio of the volume of soil water excluding nitrate to the total soil volume. Each time $y_{12}(t)$ is nonzero, a new increment is introduced, pushing all those ahead of it down a distance ΔZ.

A number of studies (*9, 12, 28*) have shown that anions, like nitrate, move with the wetting front through dry soil. The depth to the peak concentration can often be estimated by the ratio of infiltration to field capacity. Laboratory leaching studies (*21, 22*) with water contents near saturation show that these anions are excluded from some of the soil water and thus move faster than the total soil water does. Opinions differ as to the importance of this factor for field conditions. Some consider it important (*23*), others do not (*1*). We have chosen to incorporate an exclusion factor because a large part of the nitrate movement being modeled will occur at moisture contents above field capacity and it will also reflect some of the channelized flow in clay soils (*8*).

When ammonium fertilizer is applied, it is assumed to be concentrated at depth, d_1. Ammonium is converted

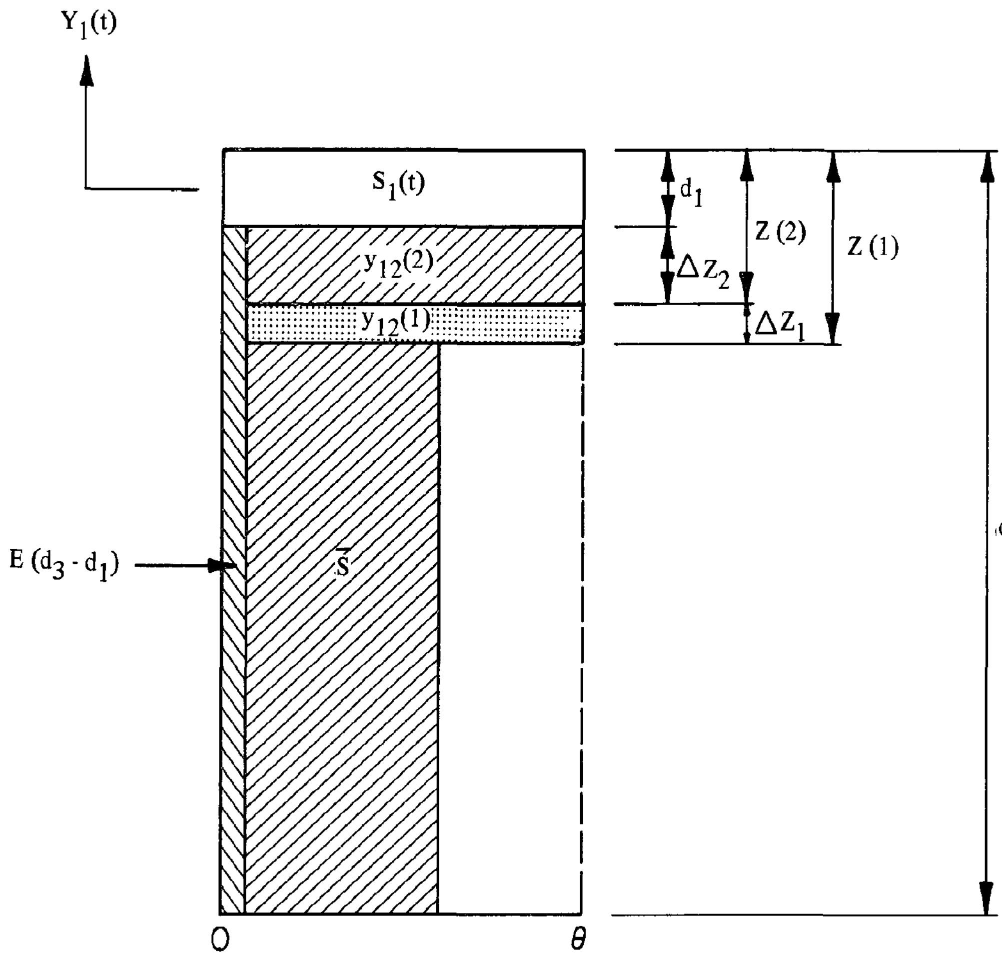

Figure 4.–Piston flow soil water model.

to the nitrate according to the following temperature-dependent relation after a lag of 5 days:

$$N(t+1) = N(t) + k(T)\,A(t) \qquad (14)$$

where N(t) is the nitrogen in the nitrate form on day (t), A(t) is the nitrogen in the ammonium form on day (t), and k(T) is a temperature-dependent rate function given by the following equations (*4, 19*):

$$\left.\begin{aligned} K &= 0.0032T - .012 \;;\; 10^\circ C \leq T \leq 35^\circ C \\ K &= 0.00105T + 0.000095T^2 \;;\; 0^\circ C \leq T \leq 10^\circ C \\ K &= 0 \;;\; T < 0^\circ \end{aligned}\right\} \qquad (15)$$

where T is the soil temperature in °C. Soil temperature was approximated by a 5-day moving average of mean daily air temperature. According to the data presented by Van Wijk et al (*26*) this appears to be a reasonable approximation.

Nitrate-nitrogen accumulates in the upper compartment until the available water storage capacity is exceeded by infiltration during a rainy day, then all of the nitrate-nitrogen is apportioned to the appropriate element ΔZ. To each ΔZ_i there is an associated weight of nitrate N, w_i. As shown in the right hand side of Fig. 5, the depths, Z_i, of each front and the weight of N, w_i, carried by each ΔZ increment are recorded and changed as each recharge event occurs. When the root zone reaches its maximum depth in the following crop year, all of the w_i below the depth d_3 are summed up to give the nitrate leaching loss from fall fertilization. The nitrate loss from spring fertilization is obtained by recording the amount of ammonium nitrogen remaining at the time of spring fertilization, and the number NR of recharge events that have occurred since fall fertilization (in Fig. 5 the value would be 4). On the date of full root

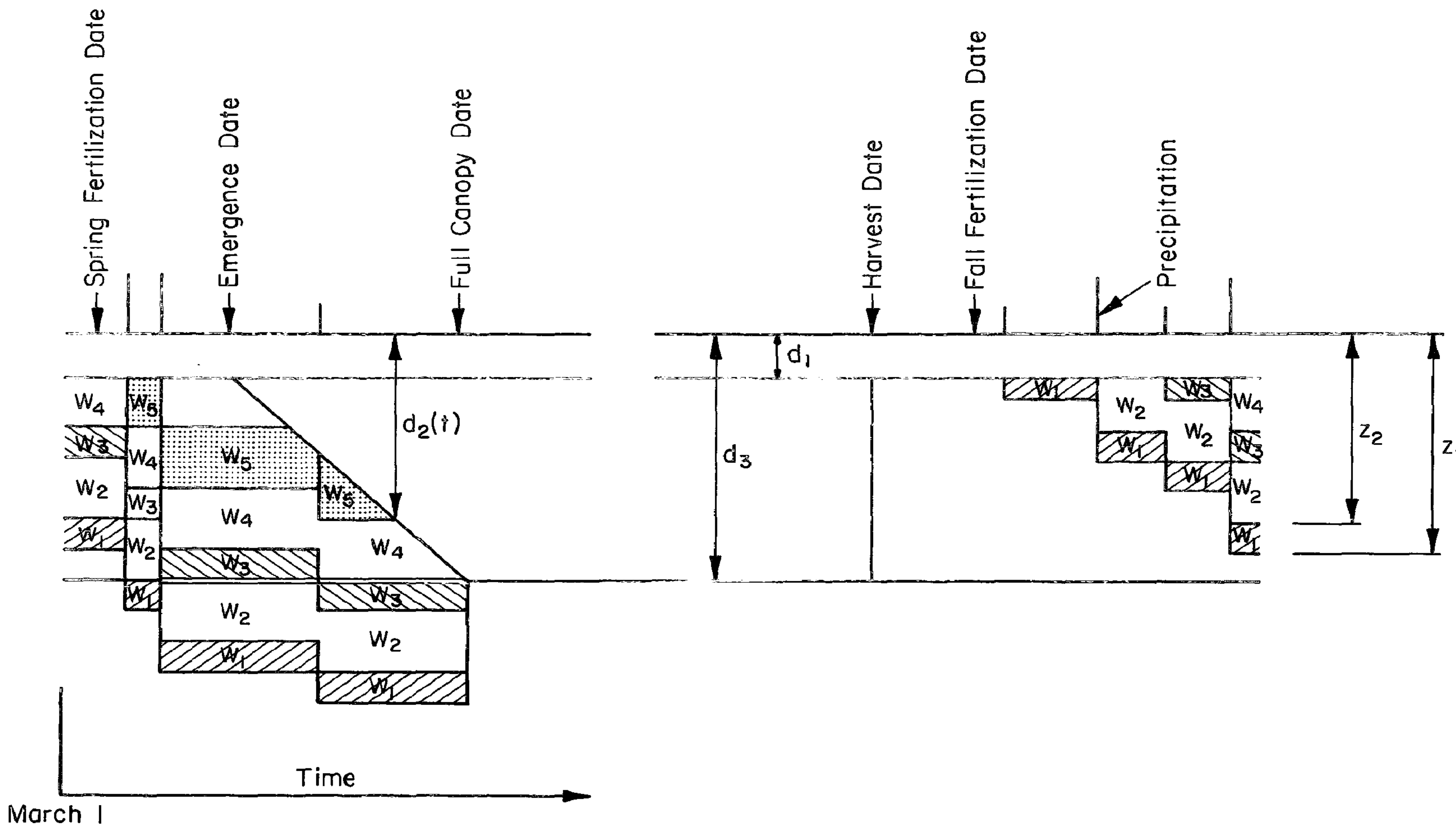

Figure 5.—Nitrate leaching model operation in time.

zone extension, all w_i from NR-1 that are below the root zone are summed and divided by the ammonium nitrogen present at spring fertilizations to give the percentage loss. Note that for the case portrayed in Fig. 5, no nitrate would be lost from spring fertilization because the first increment of recharge after spring fertilization was not below the root zone. In the piston flow model for water movement and nitrate leaching it is assumed that the nitrate is completely mixed within each element ΔZ, but that there is no diffusion or dispersion allowing nitrate exchange between elements.

SIMULATION PROCEDURE

Data

Daily precipitation and temperature data used in the simulations were obtained on magnetic tape from the National Climatic Center, Environmental Data Service, NOAA, U.S. Dept. of Commerce, Asheville, N. C. The data set obtained is termed Day Deck 345. The normal period of record was from January 1948 through December 1973. A year beginning on March 1 was used in all simulations. The stations used are listed in Table 5, Appendix A. Simulations were limited to stations east of the Rockies because of the steep rainfall gradients in the West and because most of the situations where leaching may be a problem are in irrigated areas and thus are excluded from this report.

Mean monthly pan evaporation data were obtained for the stations used or for nearby stations from a U. S. Weather Bureau publication (*25*). Fourier series were fit to these monthly values and were converted to mean daily values. A single harmonic explained more than 97% of the variance for most of the stations. The mean daily evaporation and the amplitude and phase angle of the first harmonic were plotted on maps and isolines were drawn. These parameters were then estimated by interpolation from the maps for stations where evaporation pan data were not available. The pan coefficient, K_p, was obtained from the map presented by Kohler, Nordenson and Baker (*10*).

Estimation of Parameters

The index crop considered was straight-row corn. Planting and harvesting dates for each locality were obtained from maps prepared by the USDA Statistical Reporting Service (*24*). Plowing and spring fertilization were arbitrarily assumed to have been done 14 days before planting. The fall fertilization date was the day the 5-day moving average temperature went below 50° F (10° C) or December 15, whichever occurred first. Corn was assumed to reach full canopy 80 days after planting. Root zone depths, available soil water capacities, field capacities, and exclusion fractions most commonly used in the simulations are shown in Table 1.

Table 1.—Most commonly used soil-water model parameters

Parameter	Hydrologic soil group			
	A	B	C	D
d_1	4 in.	4 in.	4 in.	4 in.
d_3[1]	4 ft.	4 ft.	4 ft.	4 ft.
K_1	.33 in.	.67 in.	.67 in.	.50 in.
$K_1+K_2+K_3$ . .	4.0 in.	8.0 in.	8.0 in.	6.0 in.
θ_F	.123	.237	.327	.345
E	.04	.07	.10	.15

[1] Root zone depths were reduced for shallow soils.

It was assumed that the available water-holding capacity of a soil was the difference between the water content at 0.3 bars and 15 bars tension. (Approximate field capacity and wilting point). Typical textures of soils in each hydrologic soil group were then selected and the total available water content was rounded to the nearest inch. The storage capacities were assigned to land resource areas on the basis of the characteristics of the predominant agricultural soils. The assignments were reviewed and corrected where necessary by soil scientists of the SCS Technical Service Centers. The assignments by land resource areas are shown in Table 6, Appendix A. The values of the exclusion fraction, E , were estimated for the assumed water-holding capacities from published data on 15 soils (*21*).

Computer Program

The subroutine ETRANS (Evapotranspiration) was written in FORTRAN IV. It is called from program SCSRO described in Appendix A. A generalized flow chart is shown in Fig. 6.

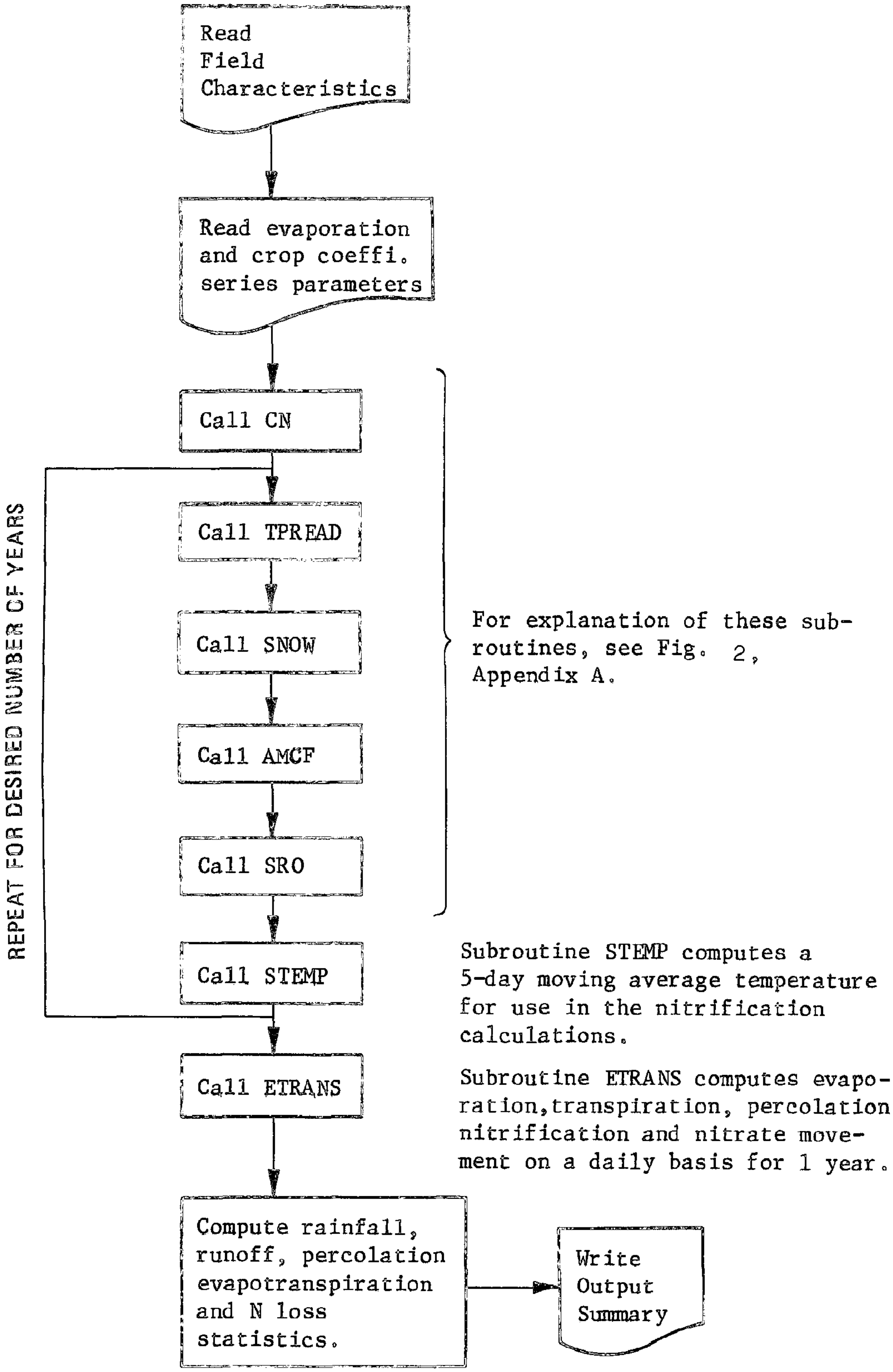

Figure 6.–Generalized flow chart. Program SCSRO with percolation and nitrate leaching option.

SIMULATION RESULTS

The program output for each 14-day period included:

1. An ordered listing of daily rainfall.
2. An ordered listing of simulated runoff.
3. An ordered listing of daily percolation.
4. The mean and standard deviation of each of the above.

The statistical summary for the n-year simulation included:

1. A table showing the number of runoff events for each 14-day period for each year.
2. The probability that there would be no runoff in any year for each 14-day period.
3. The mean annual simulated runoff.
4. The mean growing season simulated runoff.
5. Distribution functions of loss of fall-applied nitrogen, spring-applied nitrogen and leaching.
6. Mean annual percolation.
7. Mean annual evapotranspiration.

Maps of the mean annual percolation, fall-applied N loss and spring-applied N loss for each of four available soil water-holding capacities are shown in Figs. 7 through 18. Because of the limited area in which soils of Hydrologic Group A are predominant, simulations were not performed for all stations. Therefore, isolines could not be drawn in the east central portion of the United States. The mean annual precipitation for the period of record used in the simulation is shown in Fig. 19.

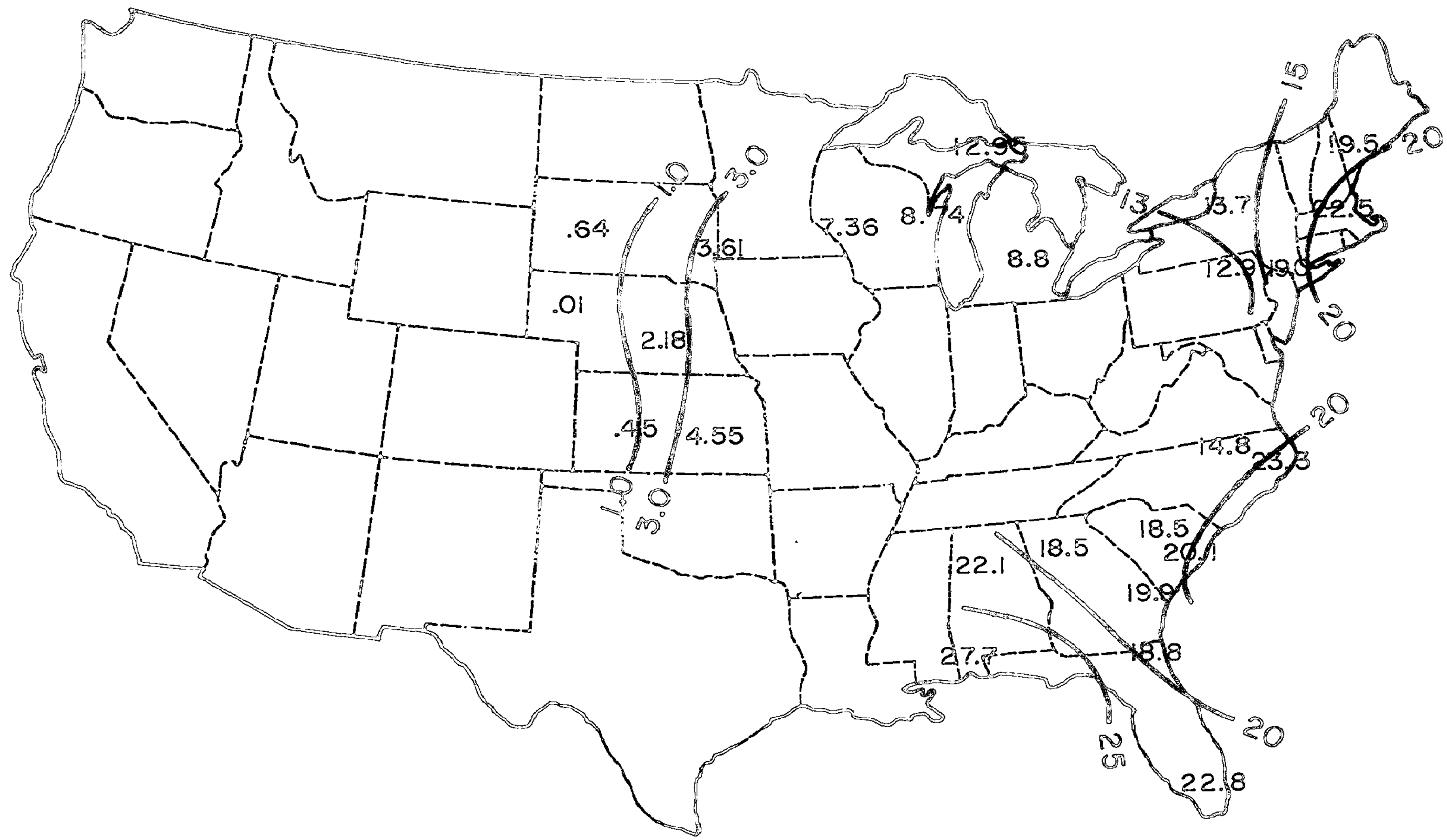

Figure 7.—Mean annual percolation below a 4-foot root zone in inches. Hydrologic Soil Group A. Four inches available water-holding capacity. Straight-row corn.

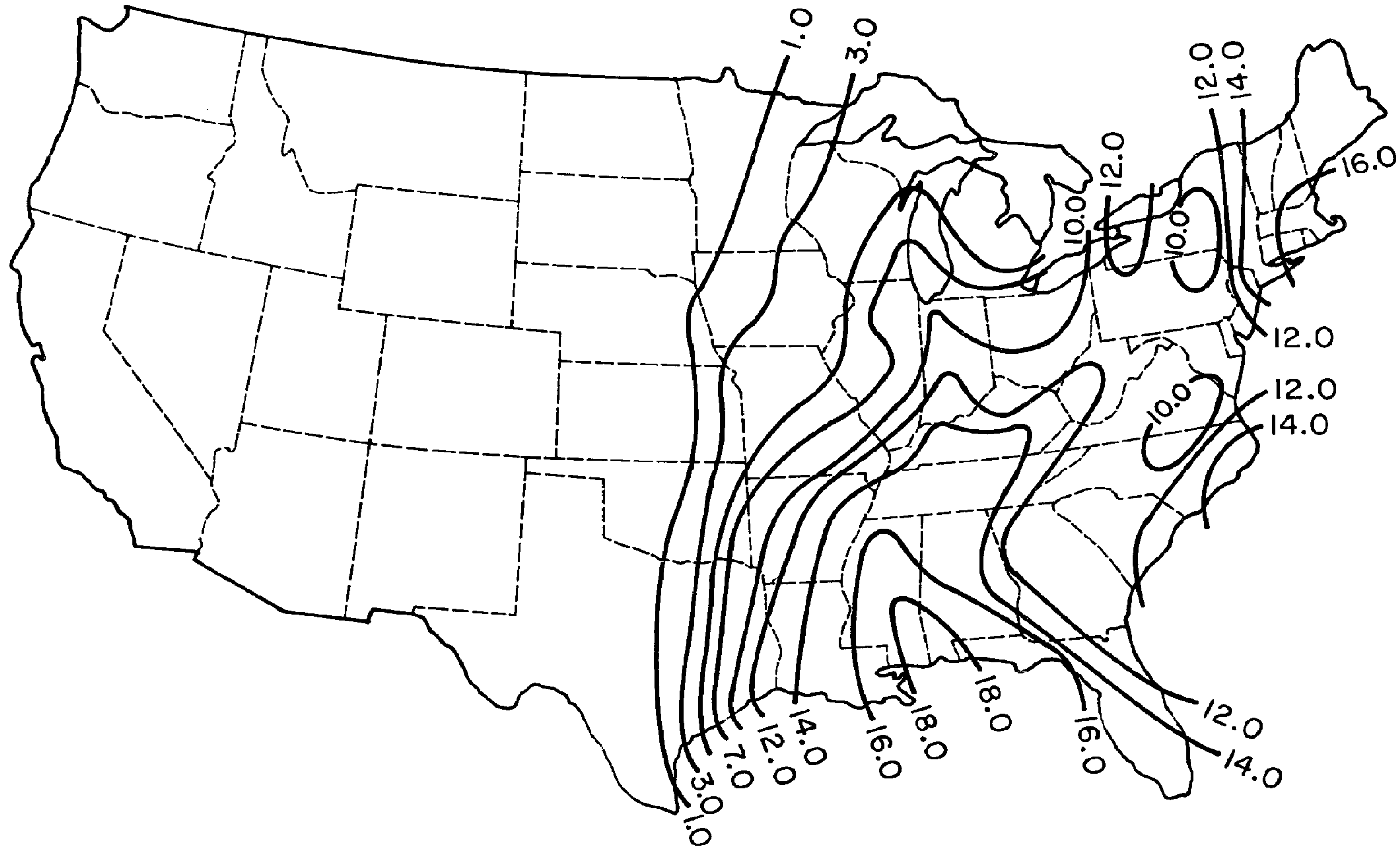

Figure 8.–Mean annual percolation below a 4-foot root zone in inches. Hydrologic Soil Group B. Eight inches available water-holding capacity. Straight-row corn.

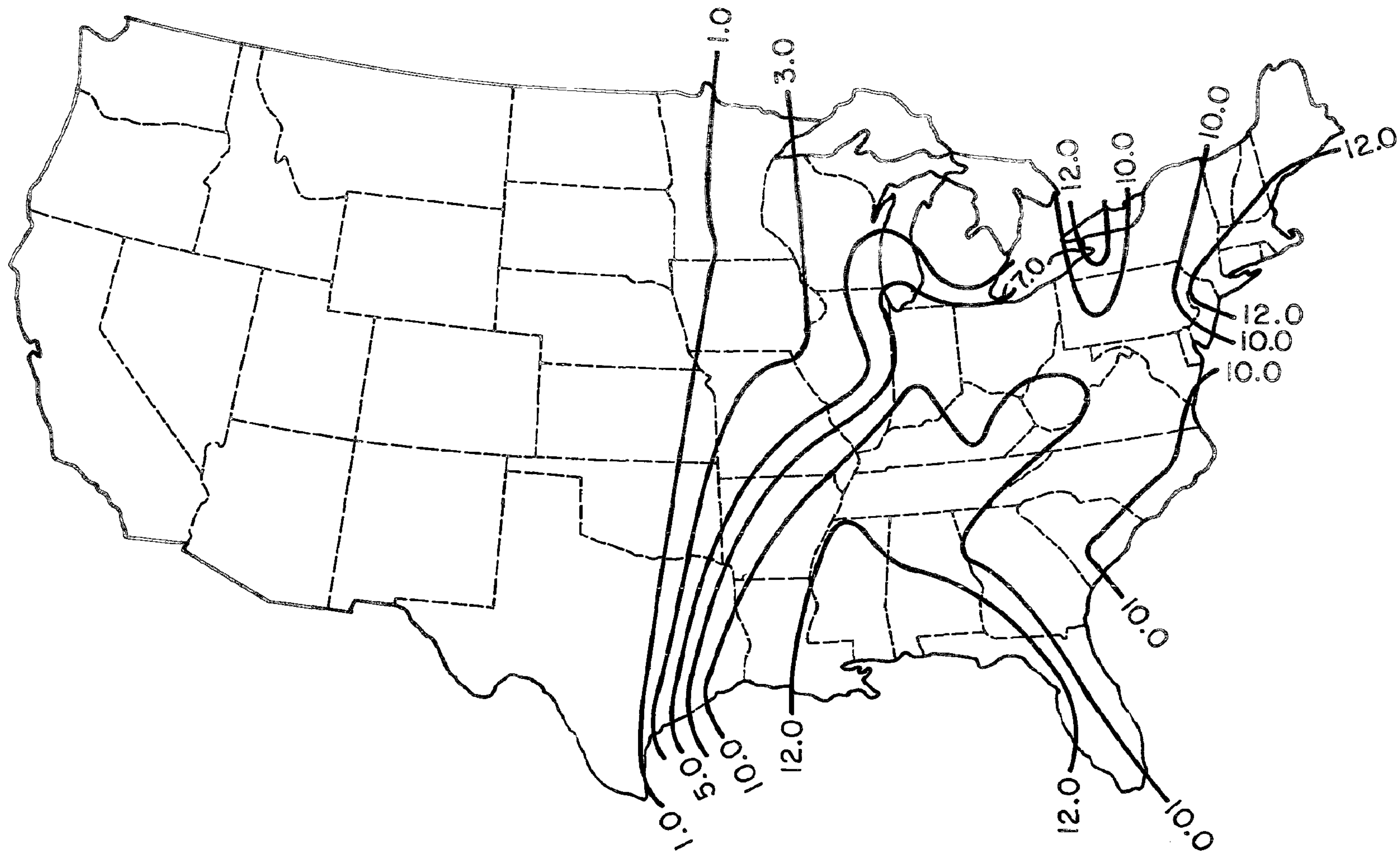

Figure 9.–Mean annual percolation below a 4-foot root zone in inches. Hydrologic Soil Group C. Eight inches available water-holding capacity. Straight-row corn.

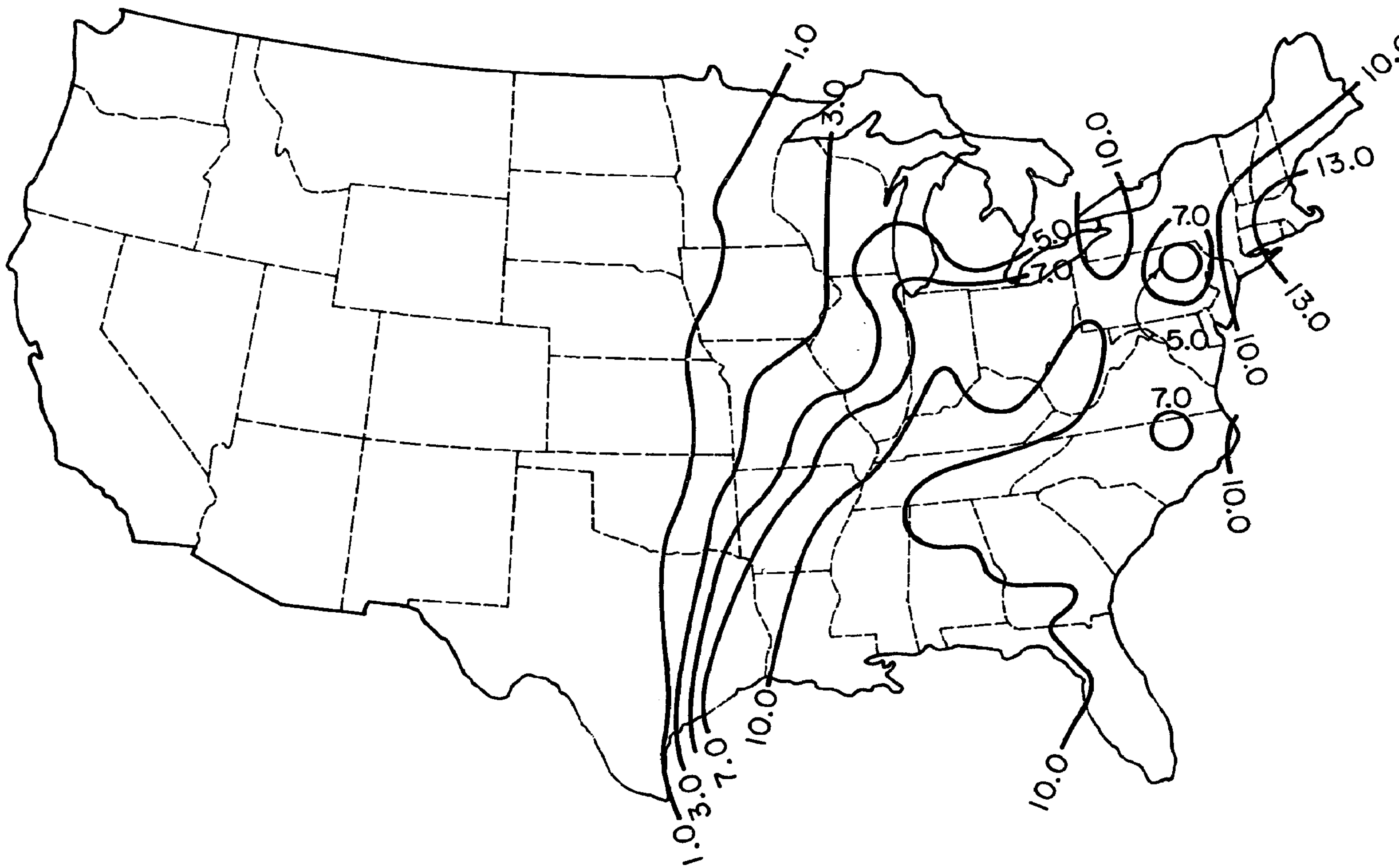

Figure 10.–Mean annual percolation below a 4-foot root zone in inches. Hydrologic Soil Group D. Six inches available water-holding capacity. Straight-row corn.

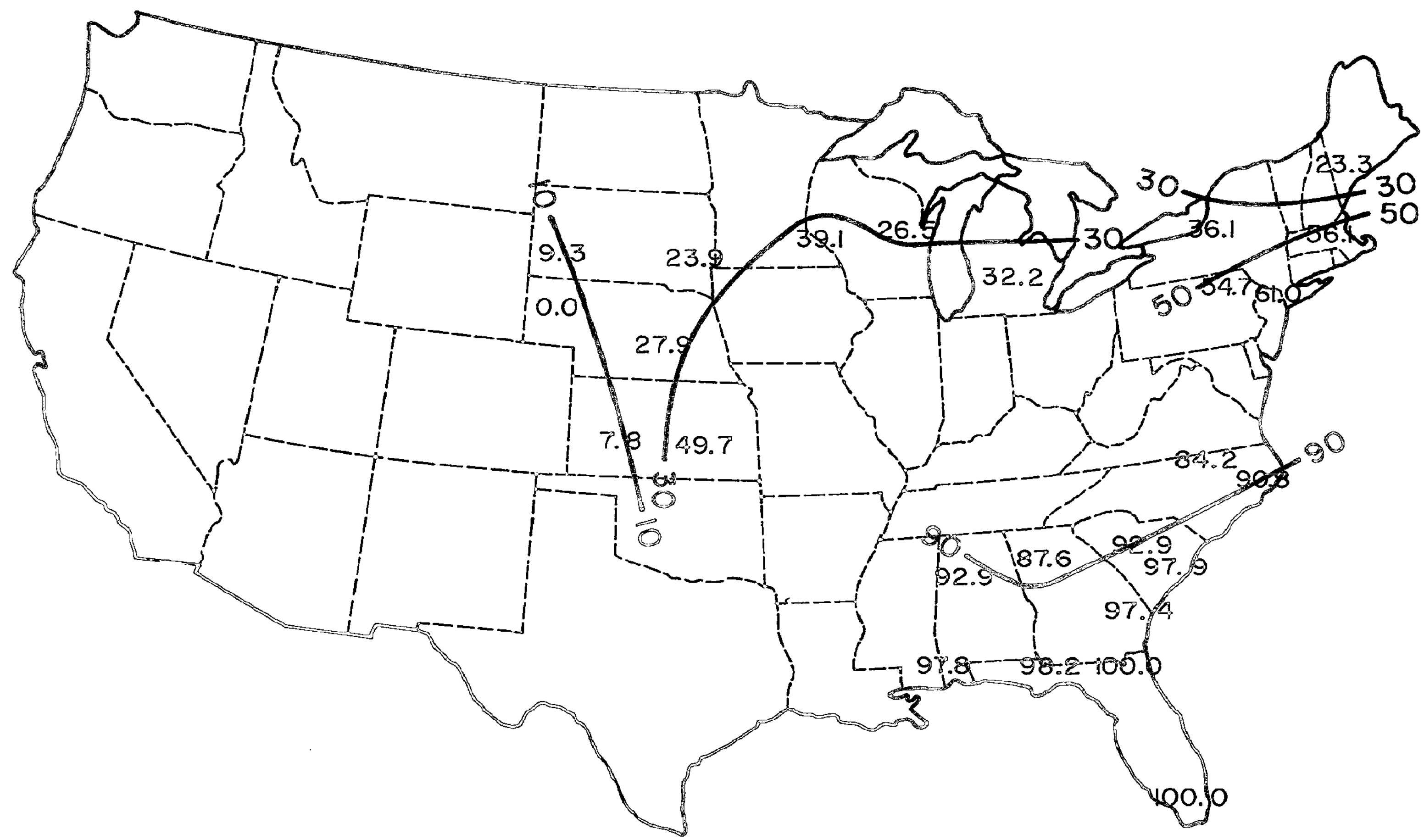

Figure 11.–Mean percentage loss of fall-applied nitrogen. Straight-row corn in good hydrologic condition. Hydrologic Soil Group A. Available water-holding capacity - 4 inches.

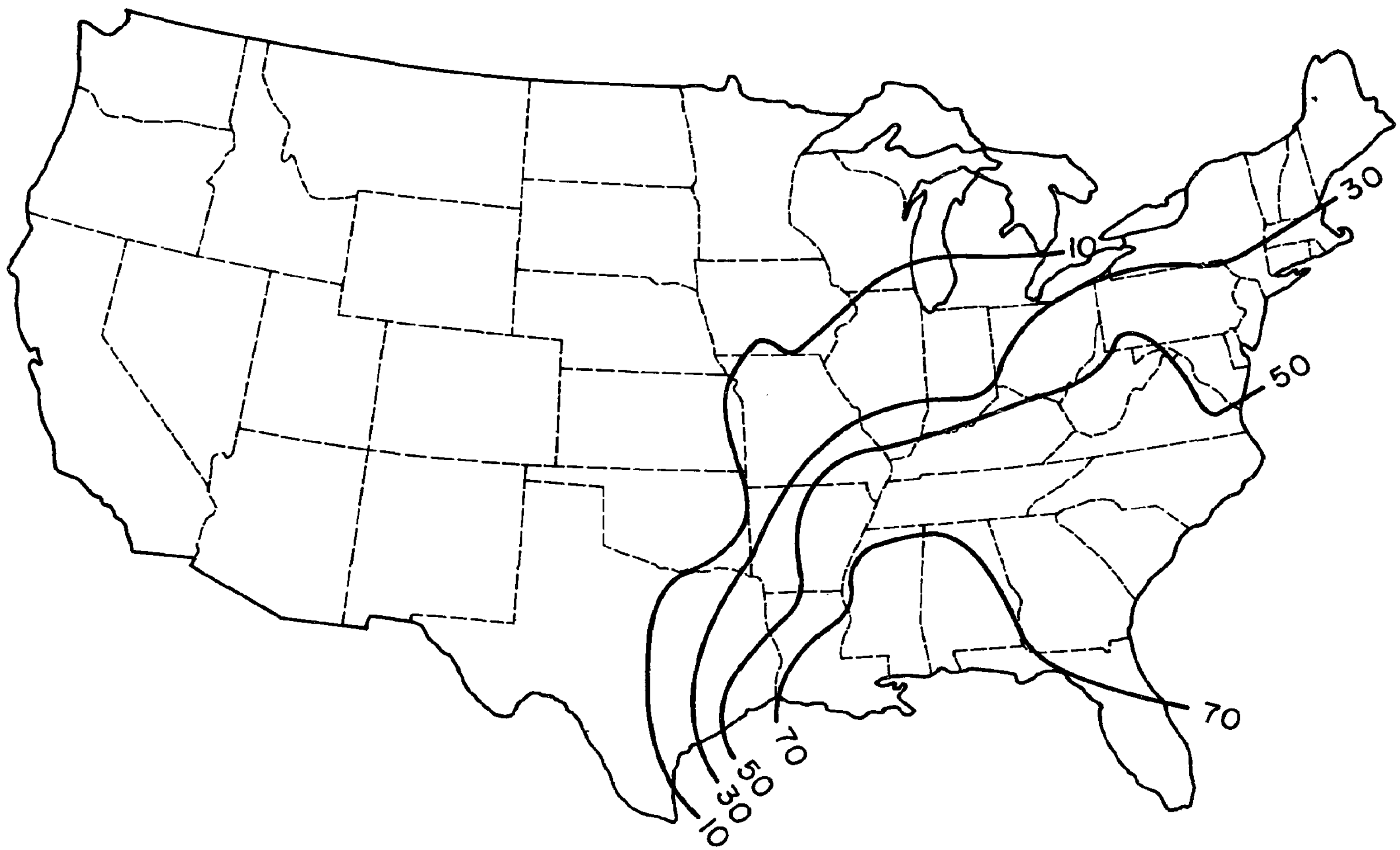

Figure 12.—Mean percentage loss of fall-applied nitrogen. Straight-row corn in good hydrologic condition. Hydrologic Soil Group B. Available water-holding capacity - 8 inches.

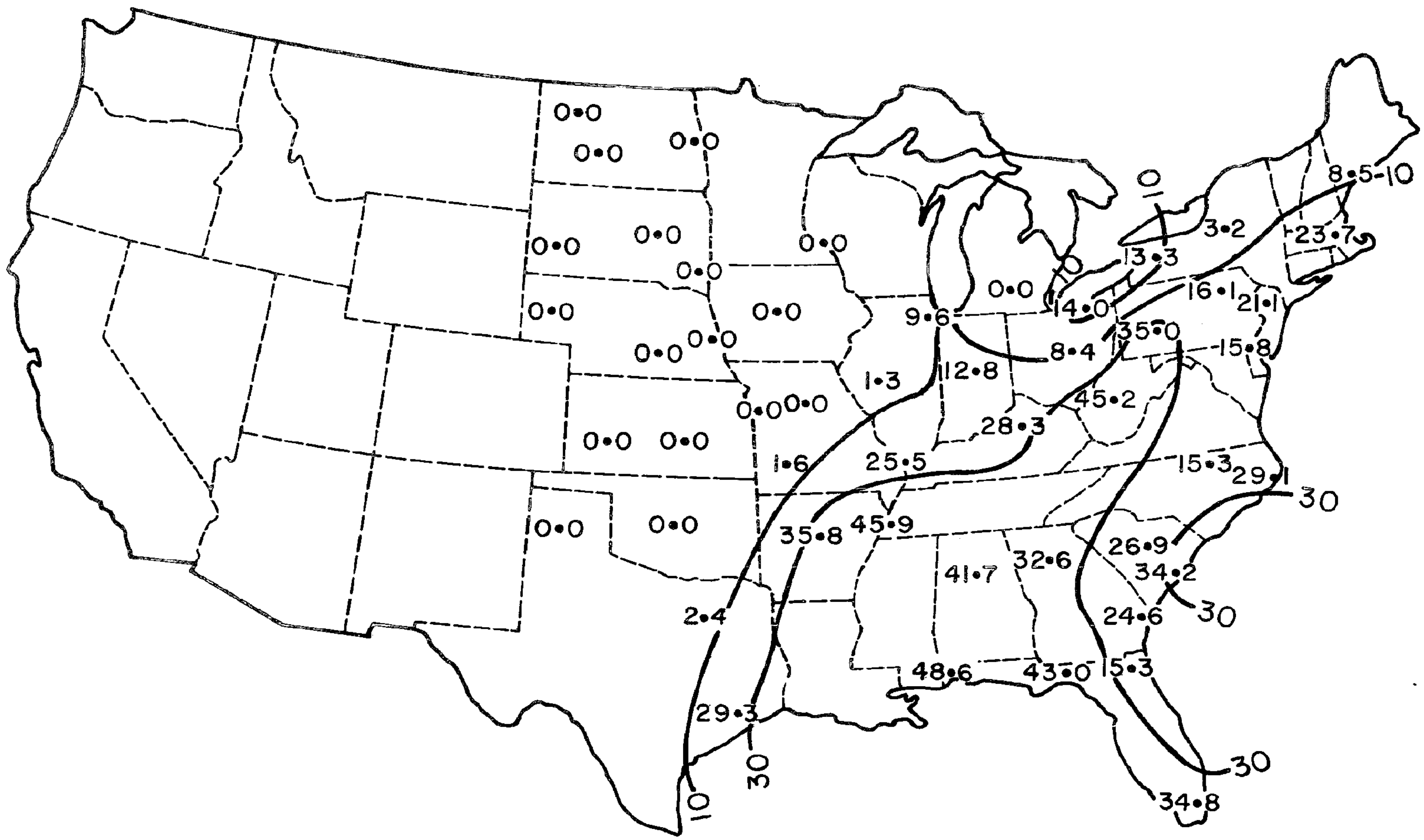

Figure 13.—Mean percentage loss of fall-applied nitrogen. Straight-row corn in good hydrologic condition. Hydrologic Soil Group C. Available water-holding capacity - 8 inches.

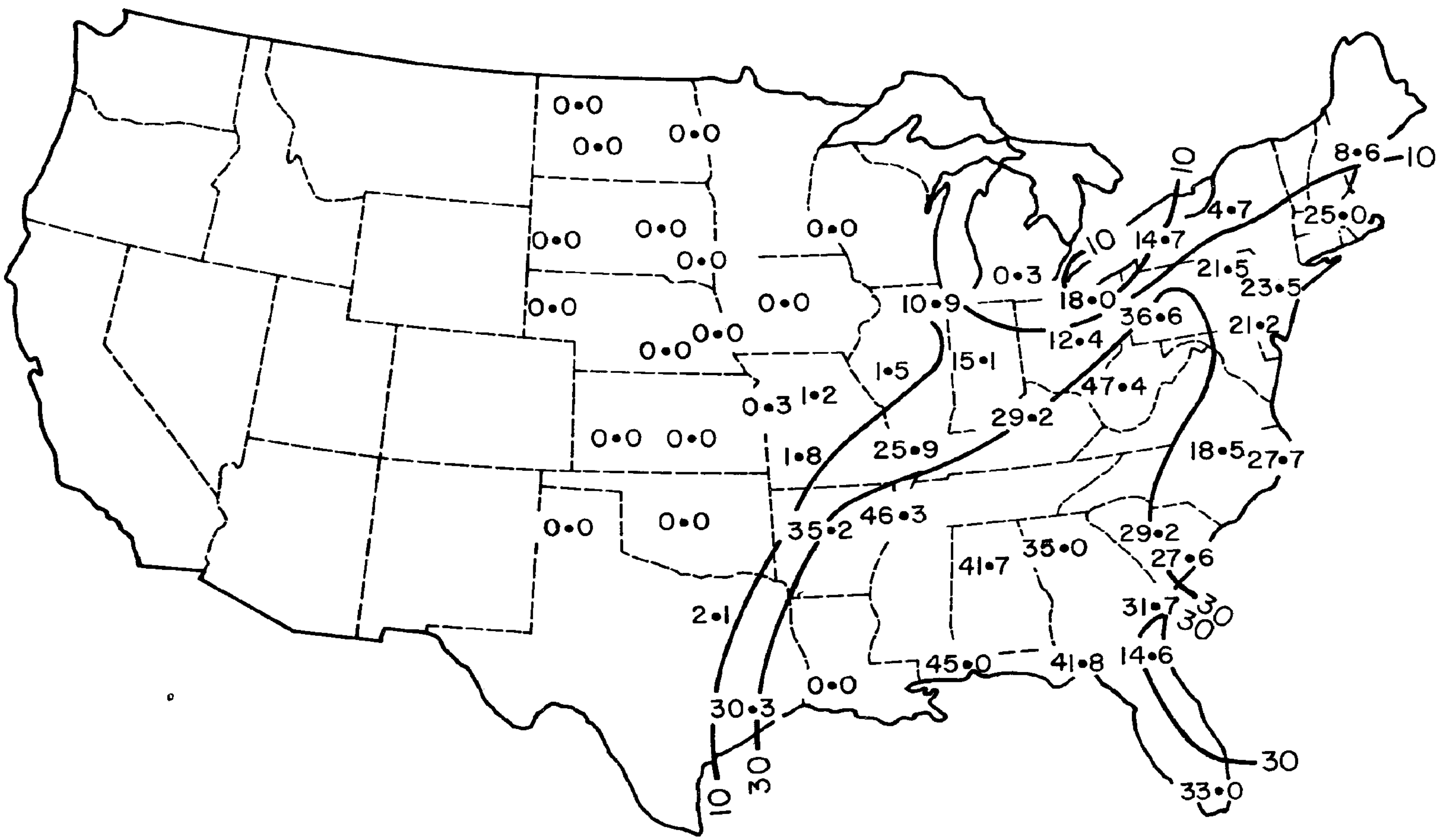

Figure 14.—Mean percentage loss of fall-applied nitrogen. Straight-row corn in good hydrologic condition. Hydrologic Soil Group D. Available water-holding capacity - 6 inches.

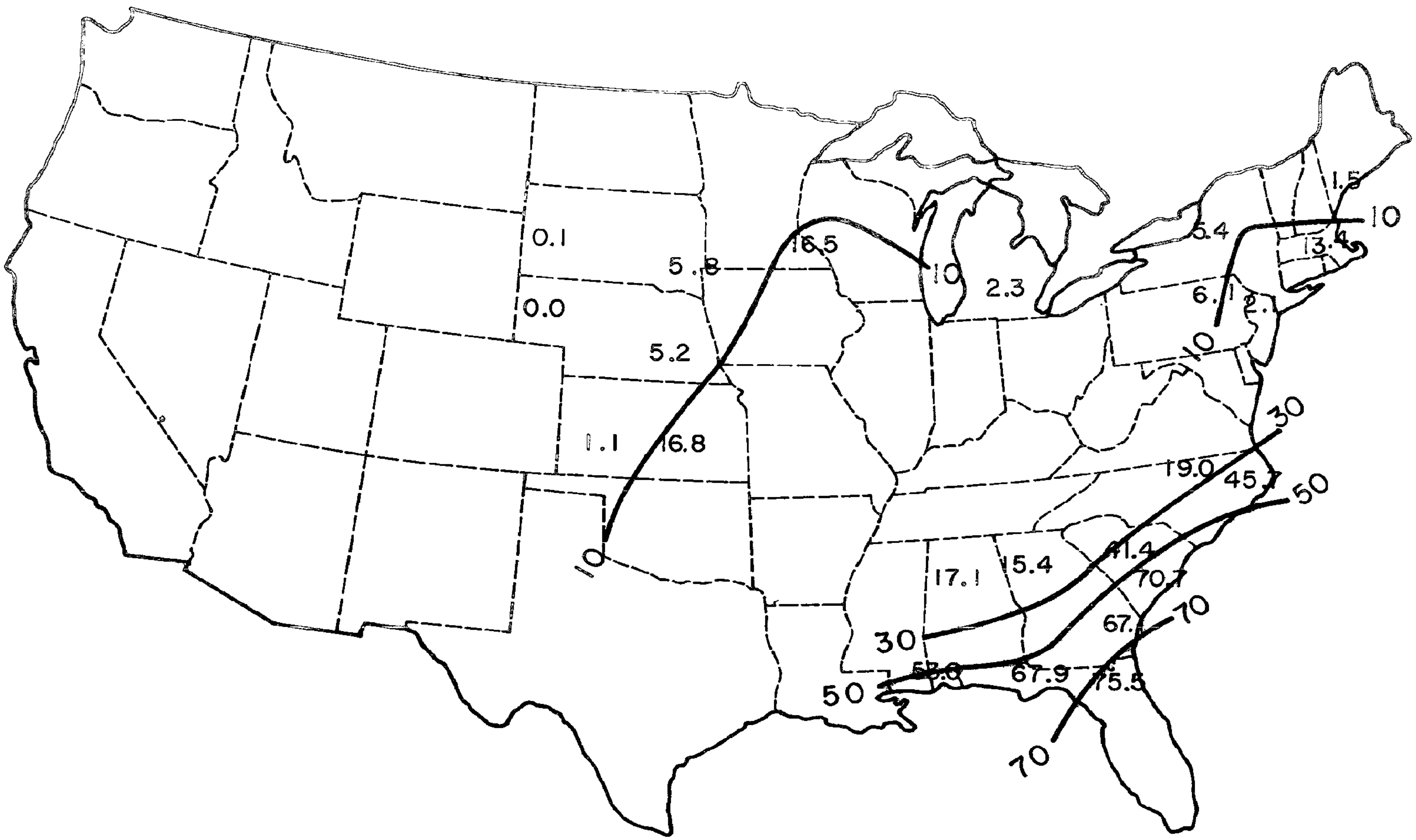

Figure 15.–Mean percentage loss of spring-applied nitrogen. Straight-row corn in good hydrologic condition. Hydrologic Soil Group A. Available water-holding capacity - 4 inches.

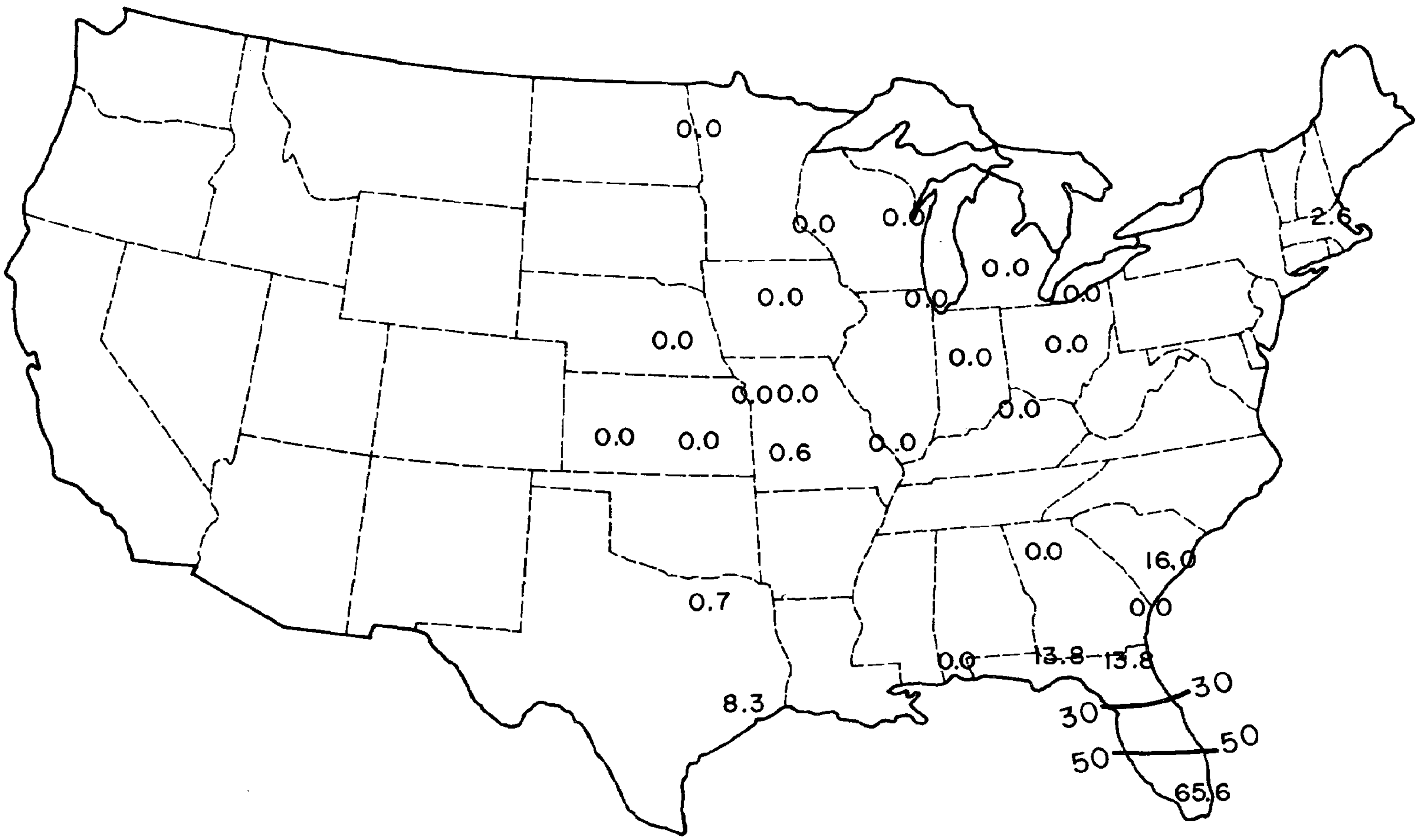

Figure 16.—Mean percentage loss of spring-applied nitrogen. Straight-row corn in good hydrologic condition. Hydrologic Soil Group B. Available water-holding capacity - 8 inches.

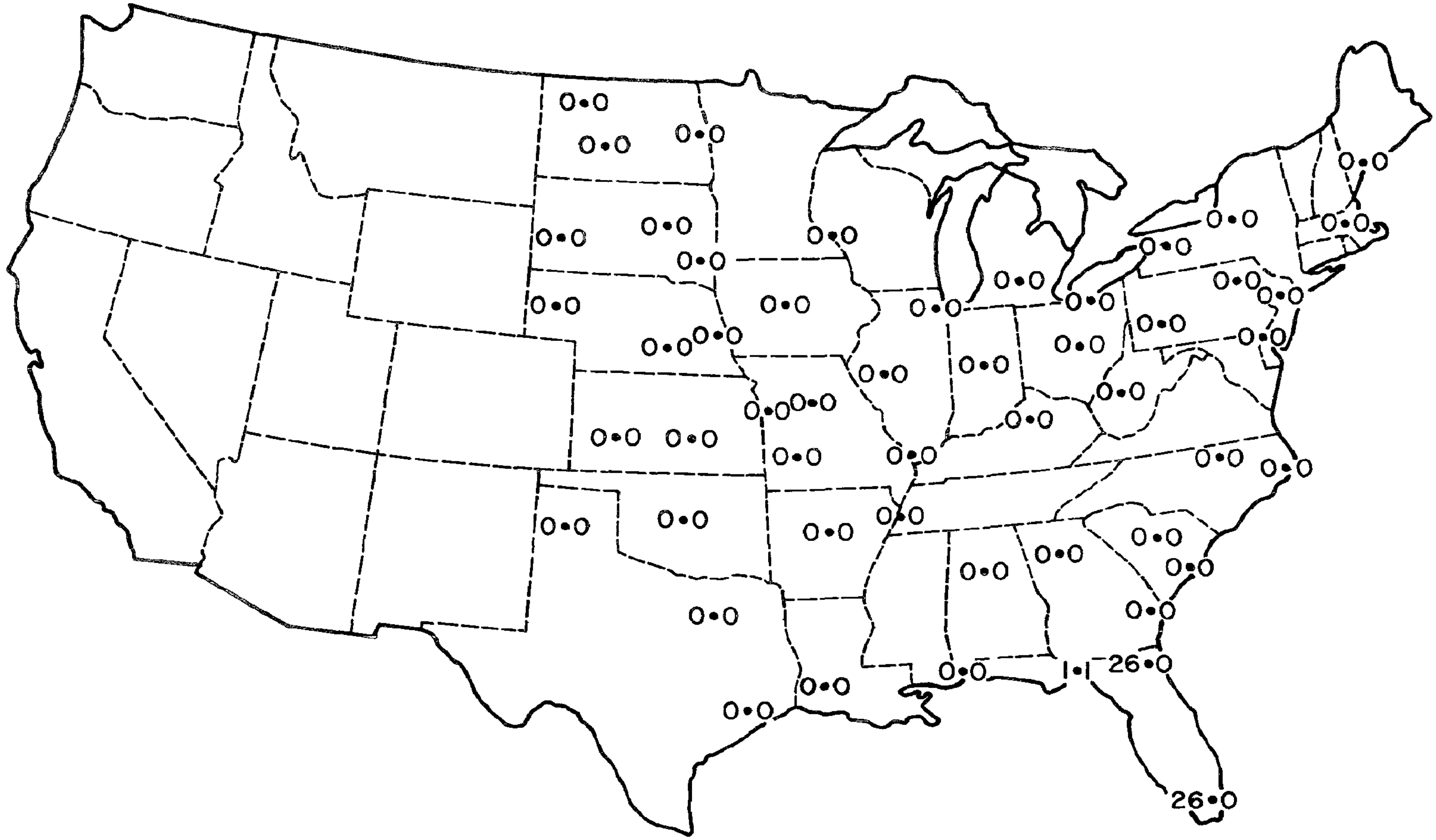

Figure 17.–Mean percentage loss of spring-applied nitrogen. Straight-row corn in good hydrologic condition. Hydrologic Soil Group C. Available water-holding capacity - 8 inches.

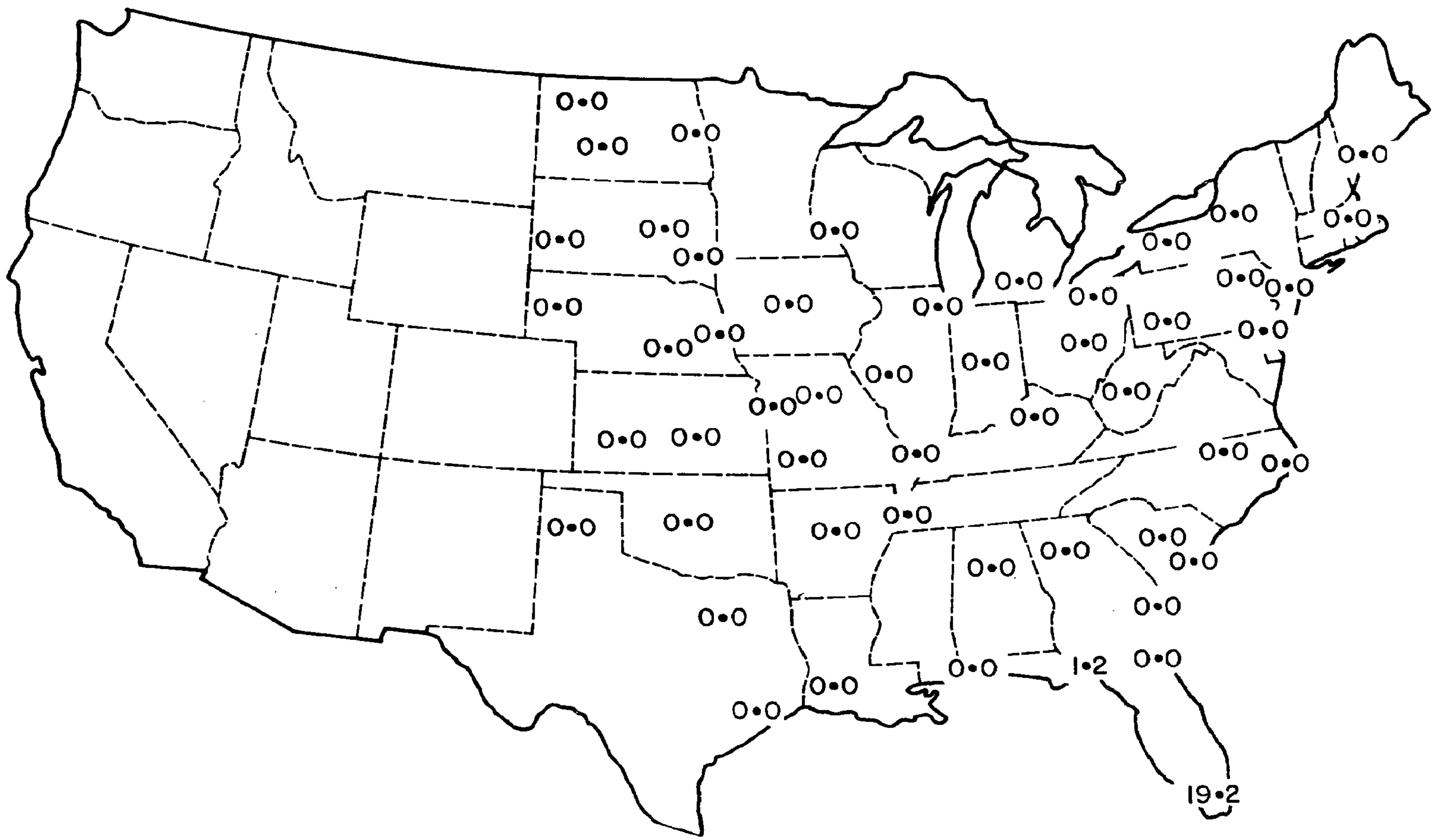

Figure 18.—Mean percentage loss of spring-applied nitrogen. Straight-row corn in good hydrologic condition. Hydrologic Soil Group D. Available water-holding capacity - 6 inches.

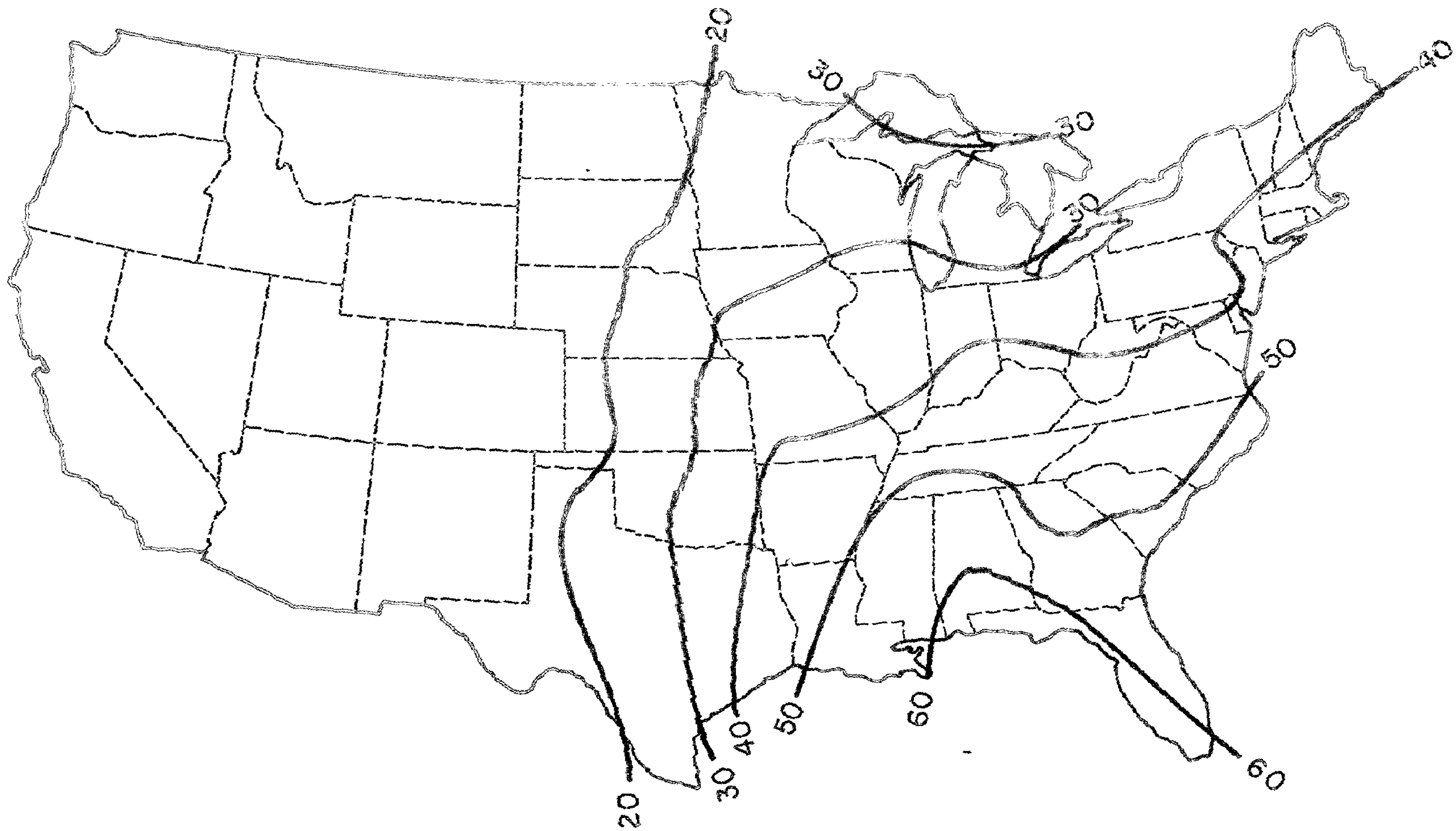

Figure 19.—Mean annual precipitation for period of record used in simulations.

DISCUSSION

Simulated mean annual percolation and nitrate leaching cannot be compared with observations because such data are not generally available. However, the excellent correlation between the sum of the simulated potential direct runoff and percolation and the surface runoff from USGS maps as presented in Appendix A suggests that the simulated results are reasonable.

Isolated bits of data are available for additional checks. Minshall (*14*) in a study of 25 years of runoff data (*1940-1964*) on the Platte River in southwestern Wisconsin found that the mean annual base flow was 5.7 inches. The mean annual potential percolation for Hydrologic Group B is somewhat greater than 5 inches (Fig. 8).

Hanway and Laflen (*6*) reported 3-year averages of 1.11 and 4.64 inches of subsurface drainage for tile outlet terraces in Creston, Iowa and Charles City, Iowa, respectively. From Fig. 8, the mean annual potential percolation for these sites is about 2 inches and 3 inches, respectively. One would anticipate greater percolation losses from tile outlet terraces than from straight row corn but the period of record is too short to make valid comparisons. Again it appears that the simulated percolation is reasonable.

Saxton, Spomer and Kramer (*20*) reported on measurements of base flow for small watersheds with contour corn near Treynor, Iowa. Six-year average base flow from two watersheds was 2.52 and 2.47 inches. From Fig. 8, simulated deep percolation in this area is about 2 inches. Rainfall during the 6-year period was above average for 5 of the 6 years.

Simulated mean annual percolation is compared with lysimeter data in Table 2. Only one set of data is for corn (Coshocton, Ohio), and the simulated percolation is very close to the observed. The other data sets agree favorably with the simulated percolation when the crop canopy differences are considered. The shallow lysimeters at Windsor, Conn. probably account for much of the difference between observed and computed percolation.

We were unable to find any data showing the percentage of fall-applied nitrogen lost during the winter and spring.

Although the comparisons between simulated percolation and data cannot be considered as conclusive, they do suggest that the simulations provide a reasonable ordering of Land Resource Areas with respect to percolation losses. The absolute amounts also appear to be realistic.

The only way a technique such as this can be judged is against readily available alternatives. The leaching hazard map prepared by Nelson and Uhland (*18*) is shown in Fig. 20. The material presented in this Appendix and in Vol. I clearly presents a more detailed picture of percolation and of the relative hazards of nitrate leaching from fall fertilization.

Care should be used in interpreting the maps of potential percolation and nitrate leaching where it is known that the model assumptions are seriously in error. For example, in the Southern United States the assumption of no nutrient uptake or transpiration during the winter would be inaccurate if winter cover crops are planted. In this case, both the percolation and the nitrate loss would be overestimated.

Table 2.–Comparison of simulated mean annual percolation with lysimeter data

Location	Citation	Soil	Hydrologic Soil Group	Crop	Average percolation: Observed	Average percolation: Simulated (corn)
					inches	inches
Ithaca, N.Y.	Bizzell (2)	Petoskey gritty sandy loam	A	Vegetables	17.76	13
			B	"	17.76	11
Geneva, N.Y.	Collison et al. (3)	Ontario, Dunkirk	B	Barley-clover rotation	12.7	11
Knoxville, Tenn.	Mooers et al. (15)	Cumberland	B	Fallow	22.5	15
Windsor, Conn.	Morgan & Jacobson (16)	Merrimac, 20" depth	A	Tobacco	13.63	20
Windsor, Conn.	Morgan, et al. (17)	Merrimac, 30" depth	A	Tobacco with winter cover	12.45	20
Coshocton, Ohio	Harrold & Dreibelbis (7)	Muskingum	C	Corn years in CWMM rotation	7.43	7

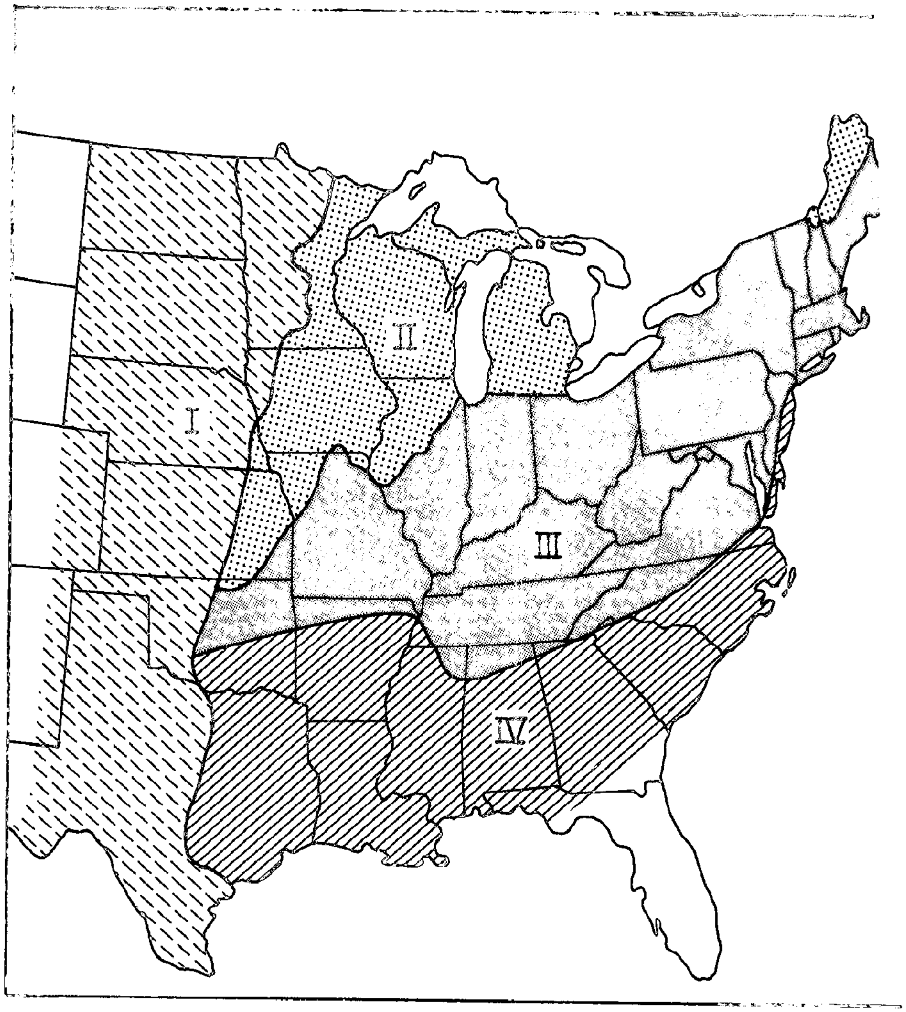

Figure 20.—Relation of degree of leaching to geographic area. Leaching ranges from nil in Area I to very high in Area IV. From Nelson and Uhland (18).

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APPENDIX C

ECONOMIC ANALYSIS METHODOLOGY

The following discussion details the application of the method presented in Section 5, Volume I, to evaluate the decision-maker's optimal choice in the example given in Section 6.2, Volume I. This example is clearly site-specific and cannot hope to show the full gamut of variables which may potentially be of significance in other situations, such as irrigation, hired labor, other crop rotations and the like. The decision-maker will have to adjust the budgeting system shown here to his particular situation.

Several important assumptions were made. For certain computations, the size of the farm became a parameter, and the assumption was made that the example farm had 250 tillable acres. Other assumptions are detailed in the following tables. In addition, it was assumed that none of the macro effects described in Section 5.2 of Volume I influence any of the decision variables noted here. This assumption implied that any machinery which may become obsolete due to a change in cropping practices would be sold at a cost close to its depreciated value and that, consequently, there was no cost of disposing of obsolete equipment to be added to the actual machinery costs.

The determination of the relevant production alternatives resulted in five potential choices, namely, (1) continuous corn no-till planted in 70 percent residue cover, contoured, (2) a corn-corn-corn-wheat-meadow rotation with moldboard plowing on the first year corn and no-till planting on the second and third year corn, contoured, (3) continuous corn with rotary strip tillage, terraced, (4) continuous corn with chisel planting, terraced, and (5) a corn-soybean rotation with no-till planting, terraced. The costs and returns for a sixth production method (i.e. continuous corn, residue left, with moldboard plowing, straight row) are shown for comparison purposes only. This particular production method does not meet the soil erosion limitation and can therefore not be considered as an available alternative.

Three of the five viable alternatives required terracing, which contributed an additional production cost, summarized in Table 1. The example assumed a farm located on Monona silt loam with more than 3 percent organic matter, a land slope of 6 percent, and an average slope length of 350 feet. According to the technical standards[1] for terrace construction, the construction of level broadbase terraces with a spacing of 120 feet would be appropriate in this situation. Table 1 shows the assumptions used in the computation of the cost of terracing the entire farm.

Table 1. Broadbase terrace construction and maintenance costs

Item	Amount
Terrace spacing, feet	120
Slope length, feet	350
Number of terraces per slope	2
Feet terrace/acre	249
Construction cost/foot terrace[a], \$.	0.60
Construction cost/acre, \$	149.40
Prorated construction cost[b], \$.	13.74
Maintenance cost, foot[a], \$	0.00023
Maintenance cost, acre, \$	0.06
Yearly terrace charge/acre, \$	13.80
Total yearly terrace charge (250 acres), \$.	3,450.00

[a] Source: Sidney James (ed), *Midwest Farm Planning Manual*, 3rd Edition, ISU Press, Ames, Iowa, 1973, p. 33.
[b] Assume 20 year life of terrace. Interest at 8 percent.

Each of these production systems requires a specific set of field operations and implements. Table 2 lists the implements considered in this study and the computation of the fixed costs for each machine. The computation of the depreciation cost used the straight-line method over the economic life of the implement. Not all of the implements were used in any particular crop production activity; Table 3 shows which implements were used in each production alternative, the total hours of machinery use, and the total implement cost. These costs did not include the cost of the tractor (listed

[1] U.S. Department of Agriculture, Soil Conservation Service, Iowa, *Technical Standards and Specifications for Conservation Practices*, Section 4A-Cropland, Work Unit Technical Guide, Code No. 600 and 602, January 1973.

Table 2. Machinery fixed costs

Machine	Size	Initial[a] cost	Salvage[b] value	Economic[c] life	Yearly depreciation	Taxes, insurance[d] and housing	Interest[e]	Yearly fixed cost
		Dollars	*Percent*	*Years*	*Dollars*	*Dollars*	*Dollars*	*Dollars*
Stalk shredder	12′ flail	2,350	13.7	12	169.00	70.50	101.83	341.33
Moldboard plow	5-16″	2,590	17.7	10	213.16	77.70	113.96	404.82
Chisel plow	15′	1,700	13.7	12	122.26	51.00	73.67	246.93
Disk, tandem	20′	4,385	17.7	10	360.89	131.55	192.94	685.38
Harrow	20′	340	17.7	12	23.32	10.20	14.73	48.25
Sprayer	tractor mounted	680	17.7	10	55.96	20.40	29.92	106.28
Planter - conventional	4-38″	1,430	17.7	10	117.69	42.90	62.92	223.51
Rotary strip planter	4-38″	3,675	17.7	10	203.45	110.25	161.70	574.40
No-till plant (fluted coulters)	4-38″	4,375	17.7	10	360.06	131.25	192.50	683.81
Wheat drill (with grass seeding attachments)	12′	2,740	9.7	14	176.73	82.20	117.43	376.36
Cultivator	4-38″	1,470	17.7	10	120.98	44.10	64.68	229.76
Duster	4-row	400	13.7	12	28.77	12.00	17.33	58.10
Combine, self-prop.	small 70-80 hp	16,100	18.9	10	1,305.71	483.00	708.40	2,497.11
Corn head	2-38″	2,800	18.9	10	227.08	84.00	123.20	434.28
Platform	13′	2,500	18.9	10	202.75	75.00	110.00	387.75
Hay mowers	7′	960	12.5	12	70.00	28.80	41.60	140.40
Hay conditioners	7′	1,300	12.5	12	94.79	39.00	56.33	190.12
Hay rake	side delivery	980	12.5	12	71.46	29.40	42.47	143.33
Hay baler	PTO	3,500	21.1	8	345.19	105.00	157.50	607.69

[a] Source: *Background Information for Use with CROP-OPT System*, FM 1628, ISU Cooperative Extension Service, Ames, Iowa, November 1974.

[b] Source: George E. Ayres, *Estimating Used Machinery Costs*, A.E. 1078, ISU Cooperative Extension Service, Ames, Iowa, January 1974.

[c] Source: Sidney James (ed), *Midwest Farm Planning Manual*, 3rd Edition, ISU Press, Ames, Iowa, 1973, Table IV-7, p. 129.

[d] Taxes and insurance at 2 percent of initial cost; housing at 1 percent of initial cost. Source: George E. Ayres, *Estimating New Machinery Costs*, A.E. 1077, ISU Cooperative Extension Service, Ames, January 1974.

[e] Assumed at 8 percent per annum.

Table 3. Machinery costs

Implement	Hours per acre[a]	Acres of use[b]	Times over[c]	Total hours	Repair cost per 100 hours[d]	Total repair cost	Yearly fixed cost	Total cost
Corn, residue left, spring turn-plow, conventional					*Dollars*	*Dollars*	*Dollars*	*Dollars*
Stalk shredder	.18	250	1	45.0	94.00	42.30	341.33	383.63
Moldboard plow......	.36	250	1	90.0	129.50	116.55	404.82	521.37
Sprayer	.21	250	1	52.5	34.00	17.85	106.28	124.13
Disk	.10	250	1	25.0	219.25	54.81	685.38	740.19
Harrow	.10	250	1	25.0	10.20	2.55	48.25	50.80
Planter	.22	250	1	55.0	114.40	62.92	223.51	286.43
Cultivator	.21	250	2	105.0	73.50	77.18	229.76	306.94
Combine	.63	250	1	157.5	322.00	507.15	2,497.11	3,004.26
Corn head	.63	250	1	157.5	56.00	88.20	434.28	522.48
Total								5,940.23
Corn, fall shred stalks, chisel plant, 30-40% residue cover								
Stalk shredder	.18	250	1	45.0	94.00	42.30	341.33	383.63
Chisel plow	.17	250	1	42.5	85.00	36.13	246.93	283.06
Sprayer	.21	250	1	52.5	34.00	17.85	106.28	124.13
Harrow	.10	250	1	25.0	10.20	2.55	48.25	50.80
Planter	.22	250	1	55.0	114.40	62.92	223.51	286.43
Cultivator	.21	250	2	105.0	73.50	77.18	229.76	306.94
Combine	.63	250	1	157.5	322.00	507.15	2,497.11	3,004.26
Corn head	.63	250	1	157.5	56.00	88.20	434.28	522.48
Total								4,961.73
Corn, residue left, strip-till row zones, 40-50% residue cover								
Stalk shredder	.18	250	1	45.0	94.00	42.30	341.33	383.63
Sprayer	.21	250	1	52.5	34.00	17.85	106.28	124.13
Rotary strip-till planter..	.22	250	1	55.0	294.00	161.70	574.40	736.10
Cultivator	.21	250	2	105.0	73.50	77.18	229.76	306.94
Combine	.63	250	1	157.5	322.00	507.15	2,497.11	3,004.26
Corn head	.63	250	1	157.5	56.00	88.20	434.28	522.48
Total								5,077.54
Corn, fall shred, no-till plant, 50-70% residue cover								
Stalk shredder	.18	250	1	45.0	94.00	42.30	341.33	383.63
Sprayer	.21	250	1	52.5	34.00	17.85	106.28	124.13
No-till planter	.22	250	1	55.0	350.00	192.50	683.81	876.31
Duster............	.21	250	1	52.5	8.00	4.20	58.10	62.30
Combine	.63	250	1	157.5	322.00	507.15	2,497.11	3,004.26
Corn head	.63	250	1	157.5	56.00	88.20	434.28	522.48
Total								4,973.11
Corn-corn-corn-wheat-meadow, residue left, no-till plant 2nd and 3rd corn								
Stalk shredder	.18	150	1	27.0	94.00	25.38	341.33	366.71
Moldboard plow	.36	50	1	18.0	129.50	23.31	404.82	428.13
Sprayer	.21	150	1	31.5	34.00	10.71	106.28	116.99
Disk	.10	100	1	10.0	219.25	21.93	685.38	707.31
Harrow	.10	50	1	5.0	10.20	0.51	48.25	48.76
No-till planter	.22	150	1	33.0	350.00	115.50	683.81	799.31
Wheat drill	.25	50	1	12.5	219.20	27.40	376.36	403.76
Duster	.21	150	1	31.5	8.00	2.52	58.10	60.62
Combine corn	.63	150	1	109.5	322.00	352.59	2,497.11	2,849.70
Combine wheat	.30	50	1					
Corn head	.63	150	1	94.5	56.00	52.92	434.28	487.20
Platform	.30	50	1	15.0	50.00	7.50	387.75	395.25
Hay mower	.31	50	3	46.5	96.00	44.64	140.40	185.04
Hay conditioner......	.31	50	3	46.5	52.00	24.18	190.09	214.27
Hay rake	.30	50	3	45.0	58.80	26.46	143.33	169.79
Hay baler	.63	50	3	94.5	210.00	198.45	607.69	806.14
Total								8,038.98

Table 3. (continued)

Implement	Hours per acre[a]	Acres of use[b]	Times over[c]	Total hours	Repair cost per 100 hours[d]	Total repair cost	Yearly fixed cost	Total cost
Corn-soybeans, no-till plant, fall shred corn stalks					*Dollars*	*Dollars*	*Dollars*	*Dollars*
Stalk shredder	.18	125	1	22.5	94.00	21.15	341.33	362.48
Sprayer	.21	250	1	52.5	34.00	17.85	106.28	124.13
No-till planter	.22	250	1	55.0	350.00	192.50	683.81	876.31
Duster.	.21	125	1	26.25	8.00	2.10	58.10	60.20
Combine corn	.63	125	1	116.25	322.00	374.32	2,497.11	2,871.43
Combine soybeans	.30	125	1					
Corn head	.63	125	1	78.75	56.00	44.10	434.28	478.38
Platform	.30	125	1	37.5	50.00	18.75	387.75	406.50
Total								5,179.43

[a]Source: *Background information for use with CROP-OPT system,* FM 1628, ISU Cooperative Extension Service, Ames, Iowa, November 1974.

[b]Acres on which implement is used each year.

[c]Number of trips through field with implement.

[d]Computed as percentage of list price. Used 2% for combine, platform, corn head and duster; 4% for stalk shredder and hay conditioner; 5% for moldboard plow, chisel plow, cultivator, sprayer, and disk; 6% for hay rake and hay baler; 8% for planters and wheat drill; 10% for hay mower. Source: George Ayres, *Estimating new machinery costs,* AE 1077, ISU Cooperative Extension Service, Ames, Iowa, January 1974.

separately in Table 4) or fuel and lubrication (Table 5). The implement hours per acre (from Table 3) were aggregated for each production alternative. The total was augmented by a 10 percent figure for traveling to field, idling, etc., to result in the total tractor hours figure listed in Table 4. The depreciation cost assumed a straight-line depreciation over the economic life of the tractor.

The fuel costs for the tractor and the combine were computed as shown in Table 5. These fuel costs were presented separately from the other machinery and tractor variable costs for the purpose of emphasizing the differences in fuel consumption among production alternatives.

Table 6 shows the computations for the seed costs of the five production alternatives. The assumption was made that the no-till alternatives would be subject to a higher seed mortality rate than the other alternatives, due to the higher crop residue levels.

Agricultural chemicals were selected on the basis of the recommended nutrient and pesticide practices. It was assumed that the nitrogen was applied in NH_3 form and the phosphate and potassium in granular bulk form (Table 7). The restrictions on optimal timing (N2) are assumed to be met by fertilizer application just prior to planting, which in the case of two alternatives (corn chisel-plant and corn rotary-strip-till) also implies incorporation (N8 and 12).

The pesticide costs (Table 8) were estimated on the basis of pesticide recommendations by the ISU Extension Service for control of the major pests.[2, 3] The pesticide costs for the several production alternatives may vary since each rotation requires the use of a unique mix of pesticides. For example, the herbicide cost for the no-till alternatives was higher than for conventional tillage because greater amounts of and more expensive types of herbicides were assumed to be used with this alternative. The insecticide cost for the rotation including meadow was assumed greater than for continuous corn alternatives due to the expected incidence of the first-year corn insect complex. No insecticide cost was assumed for soybeans, since the acreage of soybeans ordinarily treated with insecticides was quite small.

Labor costs for the five production alternatives were computed as shown in Table 9. The labor requirement per acre was estimated as 130 percent of the tractor hour requirement to account for overhead labor in addition to the direct requirements. The labor cost per hour was assumed equal to the present average wage rate for Iowa.

Table 10 presents two additional cost components, namely the corn drying costs and interest charges. It was assumed that the costs (variable and fixed) of drying corn amounted to 12 cents/bushel, which is the current charge for custom drying in Iowa.[4] It was assumed that the out-of-pocket costs involved the use of borrowed capital. The interest costs on machinery and the tractor were included in their total costs and are not repeated here.

The gross revenue for each of the production alternatives was computed as shown in Table 11. The no-till alternatives were assumed to have a slightly lower yield than the more conventional tillage alternatives due to increased production and harvesting complexities.

The final table, Table 12, summarizes all of the preceding computations and shows the gross revenue and net return figures for each of the six production methods. A land cost was included based on an assumed land value of $974.00 per acre[5] and a cash rent of $7.40 per $100 value.[6] Since this land charge applied equally to all six production methods, any error in this land charge will change only the absolute levels and not the differences in net returns among the six alternatives.

It appears that the (unavailable) alternative of continuous corn with conventional moldboard tillage has a significantly higher net revenue than any of the other (available) production alternatives. There is only a small variation in net return of the top three (available) production alternatives with a major net return drop to the corn-soybeans alternative. The corn-corn-corn-wheat-meadow rotation has by far the lowest net return among these six alternatives, indicating that the savings in fertilizer cost generated by the nitrogen nutrient credit from the legume meadow are not sufficient to offset the increases in other costs.

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[2] Harold J. Stockdale, *Insect Pest Control Recommendations for 1975,* IC-328 (Rev.), ISU Cooperative Extension Service, Ames, Iowa, January 1975.

[3] Vivan M. Jennings, *Weed Control Guide for 1975,* Pm 601 (Rev.), ISU Cooperative Extension Service, Ames, Iowa, January 1975.

[4] *Estimated 1975 Iowa Custom Rates,* ISU Cooperative Extension Service, FM 1698, Ames, Iowa, January 1975.

[5] $974.00 is the November 1, 1974 average price for high grade farmland in West Central Iowa, reported in William Murray *et al., Land Values Double in 5 years,* FM 1681, ISU Cooperative Extension Service, Ames, Iowa, January 1975.

[6] A rent of $7.40 per $100 value is the average cash rental rate for corn and soybean land reported in E. G. Stoneberg and Ronald Winterboer, *Cash Rental Rates from Iowa Farm Land,* FM 1626 (Rev.), ISU Cooperative Extension Service, Ames, Iowa, August 1973.

Table 4. Tractor costs

Item	Straight-row	Contour		Terraced		
	C conv.	C no-t.	CCCWM no-t.	C chisel	C strip	CB no-t.
Tractor hours per acre[a]	2.21	1.65	2.20	1.92	1.65	1.34
Total tractor hours[b]	607.75	453.75	605.00	528.00	453.75	368.50
Tractor initial cost,[c] dollars	18,230.00	18,230.00	18,230.00	18,230.00	18,230.00	18,230.00
Economic life, years[d]	11	13	11	12	13	14
Salvage value, percent[e]	27.5	23.5	27.5	25.5	23.5	21.5
Yearly depreciation, dollars	1,201.52	1,072.52	1,201.52	1,131.78	1,072.76	1,022.18
Taxes, insurance and housing,[f] dollars	546.90	546.90	546.90	546.90	546.90	546.90
Average annual interest,[g] dollars	795.49	785.29	795.49	789.97	785.29	781.29
Total fixed costs, dollars	2,543.91	2,404.95	2,543.91	2,468.65	2,404.95	2,350.36
Repair costs,[h] dollars	886.34	661.75	882.33	770.04	661.75	537.42
Total tractor costs, dollars (excl. fuel)	3,430.25	3,066.46	3,426.24	3,238.69	3,066.70	2,887.79

[a] Assume tractor is required for harvest hauling, in amount equivalent to time requirements for combine. Add 0.2 hours per acre for application of fertilizer with rented implements.

[b] Increased by 10 percent for idling, travel to field, etc.

[c] 100 PTO hp diesel.

[d] From Sidney James (ed.) *Midwest Farm Planning Manual*, 3rd Edition, ISU Press, Ames, Iowa, 1973, Table IV-7, p. 129.

[e] From George E. Ayres, *Estimating Used Machinery Costs*, A.E. 1078, ISU Cooperative Extension Service, Ames, Iowa, January 1974.

[f] Taxes and insurance at 2 percent and housing at 1 percent of initial cost. Source: George E. Ayres, *Estimating New Machinery Costs*, AE 1077, ISU Cooperative Extension Service, Ames, Iowa, January 1974.

[g] Assume 8 percent interest.

[h] 0.8 percent of list price per 100 hours of use. Source: Ibid.

Table 5. Fuel costs

Item	Straight-row	Contour		Terraced		
	C conv.	C no-t.	CCCWM no-t.	C chisel	C strip	CB no-t.
Total tractor hours	607.75	453.75	605.00	528.00	453.75	368.50
Fuel cost per tractor hour,[a] dollars	2.071	2.071	2.071	2.071	2.071	2.071
Tractor fuel cost, dollars	1,258.65	939.72	1,252.96	1,093.49	939.72	763.16
Total combine hours	157.50	157.50	109.5	157.50	157.50	116.25
Fuel cost per combine hour,[b] dollars	1.106	1.106	1.106	1.106	1.106	1.106
Combine fuel cost, dollars	174.20	174.20	121.11	174.20	174.20	128.57
Total fuel cost, dollars	1,432.85	1,113.92	1,374.07	1,267.69	1,113.92	891.73

[a] Fuel consumption gallons per hour = 0.044xPTO hp. Lubrication costs at 15 percent of fuel cost. Source: Sidney James (ed.) *Midwest Farm Planning Manual*, 3rd Edition, ISU Press, Ames, Iowa 1973, p. 125. Assume diesel fuel at $0.40/gal.

[b] Gasoline consumption = 2.35 gal./acre. Source: George E. Ayres, *Fuel Required for Field Operations*, AE 1079, ISU Cooperative Extension Service, Ames, Iowa, March 1974. Lubrication costs at 15 percent of fuel costs. Assume gasoline at $0.40/gal.

Table 6. Seed costs

Item	Straight-row	Contour		Terraced		
	C conv.	C no-t.	CCCWM no-t.	C chisel	C strip	CB no-t.
Corn						
Seeding rate (seeds/acre)	23,000	26,000	26,000	24,000	24,000	26,000
Assumed mortality, %	10	20	20	13	13	20
Final stand	20,700	20,800	20,800	20,880	20,880	20,800
Seed amount[a], bu.	0.274	0.310	0.310	0.286	0.286	0.310
Seed cost[b], dollars	6.85	7.75	7.75	7.15	7.15	7.75
Wheat						
Seed amount, bu.			1.5			
Seed cost[c], dollars			11.25			
Hay						
Seed amount[d], lbs			15			
Seed cost[e], dollars			24.45			
Soybeans						
Seed amount[f], bu.						1
Seed cost[g], dollars						9.50
Seed cost per acre[h], dollars	6.85	7.75	11.79	7.15	7.15	8.62
Total seed cost, dollars	1,712.50	1,937.50	2,947.50	1,787.50	1,787.50	2,155.00

[a] Based on 84,000 seeds per bushel.
[b] Assuming price of $25.00 per bushel (Iowa price, U.S. Department of Agriculture, *Agricultural Prices*. Apr. 15, 1974.
[c] Price of $7.50 per bushel (U.S. Department of Agriculture, *Agricultural Prices*, Sept. 15, 1974).
[d] Source: Sidney James (ed.), *Midwest Farm Planning Manual*, 3rd Edition, ISU Press, Ames, Iowa, 1975, p. 18.
[e] Price of $163.00 per 100 lbs. (U.S. Department of Agriculture, *Agricultural Prices*, Sept. 15, 1974).
[f] Source: Sidney James, *op. cit.*, p. 20.
[g] Price of $9.50 per bushel (U.S. Department of Agriculture, *Agricultural Prices*, Sept. 15, 1974).
[h] Average seed cost.

Table 7. Fertilizer costs

Item	Straight-row	Contour		Terraced		
	C conv.	C no-t.	CCCWM no-t.	C chisel	C strip	CB no-t.
	Pounds per acre					
Corn						
N[a]	170	170	113[b]	170	170	150[b]
P_2O_5	30	30	30	30	30	30
K_2O	20	20	20	20	20	20
Wheat						
N			60			
P_2O_5			25			
K_2O			30			
Soybeans						
P_2O_5						30
K_2O						30
Average amount[c]						
N	170	170	80	170	170	75
P_2O_5	30	30	23	30	30	30
K_2O	20	20	18	20	20	25
Cost of fertilizer per acre[d], dollars	30.90	30.90	17.06	30.90	30.90	18.38
Total cost of fertilizer, dollars	7,725.00	7,725.00	4,265.00	7,725.00	7,725.00	4,595.00
Rental of application equipment[e], dollars	187.50	187.50	125.00	187.50	187.50	125.00
Total fertilizer cost, dollars	7,912.50	7,912.50	4,390.00	7,912.50	7,912.50	4,720.00

[a] Fertilizer recommendations based on: Regis D. Voss, *General Guide to Fertilizer Recommendations in Iowa,* AG-65 (rev.), ISU Cooperative Extension Service, Ames, Iowa, August 1973.

[b] Includes fertilizer credit from meadow or soybeans.

[c] Amount per year of rotation if other than continuous corn.

[d] Assume N as NH_3 and P_2O_5 as 46 percent P_2O_5. Prices per pound are \$0.136 for N, \$0.206 for P_2O_5, and \$0.080 for K_2O. Source: Iowa price in U.S. Department of Agriculture, *Agricultural Prices,* September 15, 1974.

[e] Assume 50¢/acre for NH_3 knife and 25¢/acre for 4-ton bulk spreader.

Table 8. Pesticide costs

Item	Straight-row	Contour		Terraced		
	C conv.	C no-t.	CCCWM no-t.	C chisel	C strip	CB no-t.
Corn						
Herbicide, dollars	11.00	16.00	16.00	11.00	11.00	18.00
Insecticide, dollars	7.00	7.00	9.00	7.00	7.00	7.00
Acres	250	250	150	250	250	125
Total cost, dollars	4,500.00	5,750.00	3,750.00	4,500.00	4,500.00	3,125.00
Soybeans						
Herbicide, dollars						11.00
Acres						125
Total cost, dollars						1,375.00
Total pesticide cost, dollars	4,500.00	5,750.00	3,750.00	4,500.00	4,500.00	4,500.00

Table 9. Labor costs

Item	Straight-row	Contour		Terraced		
	C conv.	C no-t.	CCCWM no-t.	C chisel	C strip	CB no-t.
Total direct labor, hours	765.25	611.25	714.50	685.50	611.25	484.75
Overhead (30%), hours	229.58	183.38	214.35	205.65	183.38	145.43
Total labor, hours	994.83	794.63	928.85	891.15	794.63	630.18
Cost per hour, dollars	2.50	2.50	2.50	2.50	2.50	2.50
Total labor cost, dollars	2,487.08	1,986.58	2,322.13	2,227.88	1,986.58	1,575.45

Table 10. Other costs

Item	Straight-row	Contour		Terraced		
	C conv.	C no-t.	CCCWM no-t.	C chisel	C strip	CB no-t.
Corn drying			*bu.*			
Grain harvested	27,500	26,250	16,500	27,500	27,500	13,125
			dollars			
Cost per bushel	0.12	0.12	0.12	0.12	0.12	0.12
Total cost	3,300.00	3,150.00	1,980.00	3,300.00	3,300.00	1,575.00
Interest (8%) on operating capital						
Fertilizer (8 mo.)	558.98	558.98	342.19	558.98	558.98	340.32
Seed (8 mo.)	91.28	103.27	157.10	95.27	95.27	114.86
Pesticide (6 mo.)	180.00	230.00	150.00	180.00	180.00	180.00
Fuel (3 mo.)	28.66	22.28	28.54	25.35	22.28	18.75
Labor (3 mo.)	36.08	23.56	33.18	31.36	26.98	17.71
Total interest	895.00	938.09	711.01	890.96	883.51	671.64
Total other costs	4,195.00	4,088.09	2,691.01	4,190.96	4,183.51	2,246.64

Table 11. Revenue

Item	Straight-row	Contour		Terraced		
	C conv.	C no-t.	CCCWM no-t.	C chisel	C strip	CB no-t.
Corn						
Expected yield, bu./ac.	110.0	105.0	105	110.0	110.0	105.0
Area cropped, acres	250	250	150	250	250	125
Total output, bu.	27,500	26,250	15,750	27,500	27,500	13,125
Expected price, dollars/bu.	2.75	2.75	2.75	2.75	2.75	2.75
Gross revenue, dollars	75,625.00	72,187.50	43,312.50	75,625.00	75,625.00	36,093.75
Wheat						
Expected yield, bu./ac.			45.0			
Area cropped, acres			50			
Total output, bu.			2,250.0			
Expected price, dollars/bu.			4.00			
Gross revenue, dollars			9,000.0			
Meadow						
Expected yield, tons/ac.			4.0			
Area cropped, acres			50			
Total output, tons			200.0			
Expected price, dollars/ton. . . .			45.00			
Gross revenue, dollars			9,000.00			
Soybeans						
Expected yield, bu./ac.						40.0
Area cropped, acres						125
Total output, bu.						5,000.00
Expected price, dollars/bu.						6.00
Gross revenue, dollars						30,000.00
Total gross revenue, dollars.	75,625.00	72,187.50	61,312.50	75,625.00	75,625.00	66,093.75

Table 12. Summary

Item	Straight-row	Contour		Terraced		
	C conv.	C no-t.	CCCWM no-t.	C chisel	C strip	CB no-t.
	dollars					
Gross revenue.	75,625.00	72,187.50	61,312.50	75,625.00	75,625.00	66,093.75
Costs						
Tractor (excl. fuel)	3,430.25	3,066.46	3,426.24	3,238.69	3,066.70	2,887.79
Implements (excl. fuel)	5,940.23	4,973.11	8,038.98	4,961.73	5,077.54	5,179.43
Fuel	1,432.85	1,113.92	1,374.07	1,267.69	1,113.92	891.73
Seed	1,712.50	1,937.50	2,947.50	1,787.50	1,787.50	2,155.00
Fertilizer	7,912.50	7,912.50	4,390.00	7,912.50	7,912.50	4,720.00
Pesticides	4,500.00	5,750.00	3,750.00	4,500.00	4,500.00	4,500.00
Labor	2,487.08	1,986.58	2,322.13	2,227.88	1,986.58	1,575.45
Terracing	0	0	0	3,450.00	3,450.00	3,450.00
Other	4,195.00	4,088.09	2,691.01	4,190.96	4,183.51	2,246.64
Land charge (see text)	18,020.00	18,020.00	18,020.00	18,020.00	18,020.00	18,020.00
Total cost.	49,630.41	48,848.16	46,959.93	51,556.97	51,098.25	45,626.04
Net return	25,994.59	23,339.34	14,352.57	24,068.05	24,526.75	20,467.71

As the discussion in Section 5, Volume I, points out, there are a number of intangible variables not accounted for in the net return figures. One of these intangibles, that of scheduling, may be of only minor significance in this example. The scheduling of alternatives with the highest net returns does not differ sufficiently to influence the decision.

A more important consideration is the variability of yields. The variance of yield under no-till is higher than for production alternatives utilizing more tillage. This higher variance for no-till may be partly due to a lack of familiarity with this method on the part of growers. In the present example, the no-till production alternatives were assumed to have a lower yield than the other production alternatives to account for this potential yield impact. A farmer who is a risk-averter or who is utterly unfamiliar with no-till planting may be willing to accept a lower net return with a higher degree of certainty if that alternative excludes no-till planting.

One additional consideration related to the cost of terracing. The present example assumes tacitly that the full cost of terracing is borne by the farmer. Historically, society has reimbursed terracing costs through various government programs so that the farmer usually paid half the cost or even less. Under any such cost-sharing program, the relative differences in net revenue will change. In the present example, a cost-sharing program with a 50-50 split would give two of the terraced alternatives a net revenue practically identical to the (unavailable) conventional tillage continuous corn activity.

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16. ABSTRACT

Engineering and agronomic techniques to control sediment, nutrient, and pesticide losses from cropland are identified, described, and evaluated. Methodology is developed to enable a user to identify the potential sources of pollutants, select a list of appropriate demonstrated controls, and perform economic analyses for final selection of controls. The basic principles on which control of specific pollutants is founded are reviewed, supplementary information is provided, and some of the documentation used in Volume I is presented. Volume I (Report No. EPA-600/2-75-026a) is available from NTIS as report no. PB 249-517.

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